A REPORT ON THE FEASIBILITY OF HARVESTING TIMBER WITHIN BUFFER ZONES: A LITERATURE REVIEW

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HON. G. S. MERRITHEW

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Department of Natural Resources
New Brunswick

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G.C.M.

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CHAPTER 1

INTRODUCTION

The purpose of this report is to review the available literature with the intent of determining the feasibility of harvesting timber in buffer zones. Riparian vegetation has been defined as any extra-aquatic vegetation that directly influences, or interacts with, the stream environment (Meehan et al. 1977, Campbell and Franklin 1979). Throughout this report the terms "riparian zone" and "buffer zone" have been used interchangeably. The buffer zone is that part of the riparian zone normally given special consideration due to its beneficial effects on water quality and is generally expressed as a specified distance from a watercourse. Because this specified distance has been variable, and because both terms appeared to define a zone of influence on water quality parameters, it was deemed that these terms should be equivalent for this report. All widths of buffer zones presented in this report refer to only one side of the watercourse, not both.

The riparian zone is normally thought of in relation to streams and rivers but similar zones also exist around other bodies of water such as lakes, ponds and marshes. Due to their configuration and relative lack of directional flow these latter bodies of water are generally less influenced by riparian vegetation than streams and

small rivers; nevertheless, in terms of prevention of sedimentation, energetic inputs, usage by wildlife and aesthetics, the influence of riparian zones should not be discounted.

The intent of the literature review portion of this report is to present all facts, data analyses, viewpoints, alternatives and variables in an unbiased manner. This will at times make the report appear confusing or disjunct as the train of thought is not consistently going in one direction. Considerable effort will be expended to ensure that each chapter and section will be cohesive and independent but, due to the close relationship of some of the factors and variables, some overlap is inevitable. Whenever possible information from sources as geographically and climatically close to New Brunswick as possible will be collected and/or given more importance. A summary of information derived from the Nashwaak Experimental Watershed Project is presented in Appendix II.

The content and conclusions of the various information sources have been, to the best of my knowledge, cited accurately and in an unbiased manner. I take full responsibility for the interpretation and arrangement of the following information.

CHAPTER 2

EFFECTS OF THE HARVESTING OF TIMBER ON WATER

TEMPERATURE

Increases in maximum summer stream water temperature associated with removal of stream side vegetation are well documented (Reinhart and Eschner 1962, Brown and Krygier 1967, 1970, Levno and Rothacher 1967, 1969, Gray and Edington 1969, Meehan et al. 1969, Likens et al. 1970, Swift and Messer 1971, Lynch et al. 1972, Harris 1973, Swift and Baker 1973, Day and Carvell 1979, Patric 1980). The magnitude and duration of these changes in summer water temperatures have been variable (maximum daily temperature increases of 2.2 to 7.8°C) due to the large number of factors involved, including the amount of residual overhanging and surrounding vegetation to intercept solar radiation, amount of debris in and along streams, the topography (slope, aspect, valley depth), stream characteristics (depth, flow velocity, surface area), and stream channel and bank characteristics (Brown and Krygier 1967, Gray and Edington 1969, Levno and Rothacher 1969, Brown 1971, Day and Carvell 1979). The importance of each factor may also vary seasonally and diurnally; for example, topographic shading may be relatively minor near midday in summer (Brown 1970). Additional water inputs into the stream such as surface (overland), subsurface, ground water and tributary flows

will also produce variable results (Brown and Krygier 1967, Brown 1970, Day and Carvell 1979). Brown (1969, 1970) considered several of these factors in formation of his energy balance techniques for predicting the effect of stream side vegetation removal on water temperature. Streeby (1971) felt that hardwood species were more effective than softwoods in maintaining cool stream temperatures during the summer. However, Van Groenewoud (1977) found no difference in the amount of stream exposure associated with hardwood and softwood buffer strips.

The magnitude of any stream water temperature increase is usually in direct proportion to the increased exposure of the stream to incoming solar radiation (Levno and Rothacher 1969, Brown 1970, Hornbeck et al. 1975) which is a function of the amount of stream side cover remaining and the proportion of the stream exposed. The amount of energy received is a function of the season, time of day and latitude (Brown 1970). The water temperature of a relatively small stream may be little affected by riparian vegetation removal provided a narrow strip of brush and trees adjacent to the stream is reserved (Brown and Krygier 1970, Lantz 1971, Swift and Baker 1973) or the understory of brush and shrubs near the stream is left intact (Levno and Rothacher 1969, Brown 1971, Lantz 1971, Chamberlin 1982). This stream side vegetation is also important in preventing excessive amounts of silt and debris from entering the

stream. Van Groenewoud (1977) felt that a buffer strip width of 15 m or greater was sufficient in New Brunswick and that no buffer strip was needed on the north side of east-west flowing streams. Brown (1972) and Brazier and Brown (1973) indicated that canopy density was more important in determining the shading effectiveness of a buffer strip than the strip width or the volume of commercial timber within the strip. Brown and Krygier (1970) reported no change in stream water temperature in Oregon when buffer strips approximately 30 m in width were left between clearcut areas and the stream. Brazier and Brown (1973) found strips 24 m wide to provide maximum shading while strips 17 m wide could provide 90 percent (%) of this maximum.

Small streams, with large surface areas relative to the volume of water and with low flows of water during the summer, are the most affected by riparian vegetation removal (Brown and Krygier 1967, Day and Carvell 1979). It has been predicted that streams having summer flow volumes of less than 0.028 cubic metres per second (c.m.s.) (Brown 1972) and 0.0028 c.m.s. (Swift and Messer 1971) would exhibit substantial temperature increases with complete exposure to solar radiation. Van Groenewoud (1977) indicated that, although the water of many small New Brunswick streams came from cold springs, substantial increases in water temperature would still occur following stream exposure. Streams having relatively large water volumes show greater

resistance to temperature change (Day and Carvell 1979) and usually already have some degree of exposure, so any further temperature change produced by stream side vegetation removal may be less detectable (Gray and Edington 1969).

Brown and Krygier (1967) stated that solar radiation was primarily responsible for the elevated water temperatures in exposed streams and that evaporation, condensation, conduction and convection played relatively minor roles. Incoming long-wave radiation to the water is compensated for by back radiation from the water (Brown 1970). Additionally, the low wind movement usually found above the water of streams decreases the importance of convection currents (Brown 1971). This means that any increase in water temperature cannot be attributed to the warmer air associated with clearcutting (Swift and Baker 1973) and that the heat gained by the water through solar radiation will not readily be dissipated into the air. It appears that any cooling of the water of a previously exposed stream as it flows through shaded areas is due to the input of cooler tributary, surface, subsurface or ground water, rather than heat dissipation through convection, evaporation or conduction (Brown 1971, Swift and Baker 1973, Day and Carvell 1979). The degree of cooling is therefore dependent upon the amount of input water.

Brown (1969) did indicate that differences in the heat conduction of rock and gravel stream bottoms would

influence temperature prediction. Brown (1971) stated that solar energy absorbed by a solid rock bed of a small clear stream may be 15 to 20% of the incident heat and that this would tend to lower, but also increase the duration of, the maximum temperature as the bed itself would become a source of heat. Swift and Messer (1971) noted that the range of minimum and maximum stream water temperatures was increased after clearcutting near a stream in North Carolina and attributed this to the fact that the forest canopy no longer functioned to shade the stream during daylight hours or to limit radiational cooling during the night. However, they also indicated that the stream water did not substantially cool at night during periods of highest stream water temperatures. Lynch et al. (1972) indicated no changes in monthly minimum water temperatures following riparian vegetation removal in a Pennsylvania watershed. The effect of riparian cover removal on stream water temperature in winter is also conjecturable. Lantz (1971), Aubertin and Patric (1974), Reiser and Bjornn (1979), Chamberlin (1982) and Everest and Harr (1982) have suggested that removal of stream side cover may result in lower winter and spring water temperatures. Reinhart and Eschner (1962) noted lower minimum water temperatures on a clearcut watershed in West Virginia. However, Meehan et al. (1969) reported little effect on winter stream water temperature in Alaska and Likens et al. (1970) and Swift and Messer (1971) found higher than normal stream

water temperatures during the winter in New Hampshire and North Carolina, respectively. Likens et al. (1970) speculated that the increase in winter maximum water temperatures they observed could have been due to greater snow depth, and hence greater insulation, in the newly exposed stream channel.

The duration of water temperature increases associated with stream exposure is dependent on the rate of recovery of stream side vegetation. Vegetative recovery sufficient to lower the water temperature of small streams has been reported to be fairly rapid, usually 4 to 6 years (Brown and Krygier 1970, Swift and Messer 1971, Patric 1980).

STREAM FLOW

It is not the intent of this report to review in detail the hydrological effects of vegetation removal on the water yield from forested watersheds. Nevertheless, it is felt that this aspect should be considered in assessing the potential effects of riparian vegetation removal as any management (e.g. harvesting of trees) involving the riparian zone will normally be accompanied by treatment of the entire watershed. Additionally, potential effects of riparian management may be alleviated or enhanced by hydrological changes associated with management of the entire watershed. A considerable wealth of information is available on this topic and several reviews have been prepared (Trimble et al. 1963, Hibbert 1967, E.A. Johnson 1967, Lull and Reinhart 1967, Jeffrey 1968a, Lull 1970, Rothacher 1970a, Hillman 1971, Tryon 1972, Harr et al. 1982).

The general consensus of watershed studies (involving clearcutting, patch cutting and various types of selection cutting) from Oregon (Rothacher 1970a, Harr et al. 1975, 1979), Alberta (Hillman 1971, Swanson and Hillman 1977), Minnesota (Verry 1972), New Hampshire (Hornbeck et al. 1970, Hornbeck 1973, Bormann et al. 1974), North Carolina (Hewlett and Helvey 1970), West Virginia (Reinhart and Eschner 1962, Reinhart et al. 1963, Patric and Reinhart 1971, Aubertin and Patric 1974), Pennsylvania (Lynch et al. 1972) and California (Rice and Wallis 1962) is that removal

of forests results in increased water yield and that the amount of yield is generally in direct proportion to the amount of removal. As a consequence the number of days of low flow (water flow below a predetermined minimum limit) is reduced. The size of cut area necessary to detect stream flow increases in the above mentioned studies has been variable but increases have been reported for clearcut areas as small as 8.5 ha (Lynch et al. 1972). Water yield increases following removal of riparian vegetation have also been reported (Dunford and Fletcher 1947, P.B. Rowe 1963, E.A. Johnson 1967, Lull and Reinhart 1967, Campbell 1970, Patric and Reinhart 1971, Horton and Campbell 1974).

Not all studies, however, have produced similar results. In Oregon, water yields have remained constant or actually decreased following experimental cutting, possibly attributable to reduced fog interception and subsequent drip (Harr 1980) or short-term snow melt patterns (Harr and McCorison 1979). Indeed, the effects of forest vegetation removal in a large watershed may be negligible, depending on the amount of removal (Harr 1976).

Water yield increases associated with cutting are generally attributed to a reduction in evapotranspiration [reduction in interception, and subsequent evaporation losses, and reduction of moisture withdrawal from the soil through transpiration (Lull and Reinhart 1967, Hornbeck et al. 1970, Hillman 1971, Patric and Reinhart 1971, Harr et

al. 1975, 1979, Chamberlin 1982)]. Evaporation losses from the soil after cutting are considered to be much less than evapotranspiration losses (Trimble et al. 1963). The magnitude and timing of these increases in water yield have been variable, being dependent on several factors unique to any particular area. These include the climate, drainage patterns, geology, soil type and depth, amount of deep seepage, species present (variation in root structure, surface area, length of dormant season) and treatment methods (Chapman 1962, Trimble et al. 1963, Douglass 1967, Goodell 1967, Holstener - Jorgensen 1967, Swank and Miner 1968, Hornbeck et al. 1970, Rothacher 1970a, Swanson and Hillman 1977). Increases have been commonly reported during the growing season as this is the time of maximum evapotranspiration (Reinhart and Eschner 1962, P.B. Rowe 1963, Hornbeck et al. 1970, Patric and Reinhart 1971, Lynch et al. 1972, Harr et al. 1979). However, the water content of the soil is often below maximum retention capacity during the growing season and water yield increases may not show up until the late summer or fall soil moisture recharge period (Reinhart et al. 1963, Lull and Reinhart 1967, Rothacher 1970a, Harr et al. 1975) or even until the following spring during snow melt (Reinhart et al. 1963, E.A. Johnson 1967, Swanson and Hillman 1977).

The higher moisture content of the soil due to reduced evapotranspiration results in less capacity for

storing additional water and thus an earlier increase in water yield (stream flow) over the annual cycle. Furthermore, the soil is more responsive (in terms of increased water yield) to precipitation (Trimble et al. 1963), resulting in quicker and higher average peak stream flows following storm events. Increased peak flows may result in erosion of stream banks and beds (Harr et al. 1979). However, maximum storm flows have generally been found to be unaltered by cutting, provided soil disturbance is minimal, as the soil moisture in both disturbed and undisturbed watersheds is usually fully recharged during these events (E.A. Johnson 1967, Lull and Reinhart 1967, Rothacher 1973, Harr et al. 1975). In areas of year round soil moisture deficits, such as reported in North Carolina (Hewlett and Helvey 1970), storm flow increases associated with clearcutting would also be expected year round. However, if soil moisture during the dormant season is equal on both logged and unlogged watersheds then runoff and storm flows should be equal (Patric and Reinhart 1971, Bormann et al. 1974).

Quick storm flows are, however, directly related to the amount of overland flow, which is more rapid than ground water flow (Harr et al. 1975). The amount of overland flow is greatly influenced by the degree and continuity of soil disturbance and compaction, and the interception of subsurface flows by road cuts and ditches, as well as the soil and watershed properties (Reinhart et al. 1963, Harr et

al. 1975, 1979, 1982, Swanson and Hillman 1977). Sheridan et al. (1967) reported little increase in the magnitude of peak flows following logging in Alaska and attributed this to the thickness of the organic material, which allowed for rapid perculation of water through the soil, and low transpiration rates (prior to logging) during heavy rainfall periods. Additionally, the prohibitive costs of road building kept the road network to a minimum. Any increase in the amount of surface runoff would advance the timing of peak flow events following storms. Low flows may be reduced due to the reduction in soil moisture storage caused by this increase in surface runoff; however, large subsurface water channels (such as root channels) may be only partially restricted in some cases, resulting in slower water movement through the soil and a delay in peak flows (Chamberlin 1982). Thus, the amount and timing of stream flow peaks associated with watershed harvesting appears to be variable, which makes prediction difficult. Chapman (1962) emphasized the difficulty in predicting peak stream flow events, as well as the amount of stream flow for any particular year, and attributed this to the interaction of precipitation, transpiration, and infiltration rate, compaction and retention capacity of the soil.

Goodell (1967) concluded that, unless slopes were steep, aspect would have little effect on changes in water yield as the orientation of tree crowns tended to equalize

the amount of solar radiation reaching the canopy. In contrast, Hibbert (1967) found northerly aspects to have larger increases than southerly aspects on slopes ranging from 26 to 46% in North Carolina. He attributed this to differences in evapotranspiration but gave no reason for the direction of the trend.

Water yields gradually return to normal as new growth develops (Lull and Reinhart 1967, Berndt and Swank 1970, Rothacher 1970a, Swanson and Hillman 1977, Harr et al. 1979). The length of time over which this process occurs is highly variable depending on the extent of original disturbance and the rate and type of reforestation, but the effects of cutting on water yield have been recorded for up to 30 years (Swanson and Hillman 1977, Harr et al. 1979).

The timing of any stream flow increases associated with the cutting of watersheds in New Brunswick would likely be variable, and thus difficult to predict, depending on the moisture content of the soil and the variety of previously mentioned factors. Lull and Reinhart (1967) concluded that cutting in the northeastern United States would increase spring flows due to reduced snow interception. Midwinter stream flows may be reduced as most precipitation is in the form of snow and melting is reduced (Hornbeck 1973), and frost penetration is deeper in cutover areas (Verry 1972). Even in undisturbed areas of the northeast, winter runoff is

small while the spring runoff, following snow melt, is relatively large (Sopper and Lull 1965).

SNOW ACCUMULATION

Snow accumulations have been consistently found to be higher than average in forest openings (Berndt 1965, Miller 1966, Stanton 1966, Clausen and Mace 1972, Gary 1974, 1975, 1979). Originally this was considered to be due to reduced interception of snow and its subsequent evaporation from the canopy (Gary 1975). It now appears that alteration of wind patterns in small openings is responsible for the larger amounts of snow, in effect reducing the amount of snow under the adjacent uncut canopy (Miller 1966, Stanton 1966, Gary 1974, 1975, 1979, Chamberlin 1982). Thus, the total amount of stream flow associated with snow melt remains relatively constant following cutting (Hornbeck and Pierce 1969, Verry 1972) as only the snow distribution is changed. Any increased spring season water yields following cutting in a watershed are likely the result of increased soil moisture content due to reduced evapotranspiration losses the previous summer rather than a reduction in snow interception - evaporation losses from the canopy (Jeffrey 1968b). Chamberlin (1982) felt that small openings (up to 8 tree heights) in the forest canopy were more effective in accumulating additional snow amounts than larger ones although Hillman (1971) suggested that increased snow accumulations would also occur in large clearcut blocks in Alberta. The size of openings in the above mentioned studies was variable, with strips ranging from 10 to 150 m

wide and patches ranging from 2 to 8 ha. An inverse relationship between canopy density and snow accumulation has also been suggested (Berndt 1965).

Snow melt has been reported to be earlier and of a higher rate in forest openings (Berndt 1965, Stanton 1966, Hornbeck and Pierce 1969, Anderson 1970), attributable to less shading and greater wind movement, allowing convective heat transfer (Anderson 1970). Greater soil moisture content in openings may also cause earlier and higher peak snow melt flows (Chamberlin 1982). This may result in a desynchronization of spring melt from partially cut watersheds (Hornbeck and Pierce 1969, Hornbeck et al. 1970, Verry 1972, Chamberlin 1982), reducing maximum flow volume and flooding potential. However, synchronization of snow melt flows may also occur if advanced snow melt from openings created on upper elevations or on north-facing slopes coincides with the normal snow melt from warmer portions of a watershed (Chamberlin 1982). Flooding potential is also a consideration after cutting large blocks in Alberta (Hillman 1971).

It must be emphasized that snow accumulation and melt in forest openings are dependent upon many factors such as the size and shape of openings, amount of slash present, aspect, elevation and forest canopy [see Jeffrey (1968b), Meiman (1968), Anderson (1970) and Chamberlin (1982) for a review of these factors as well as the effects of radiation

and convection currents on snowpacks]. Slash may increase the rate of snow melt by absorbing solar radiation, which is subsequently transferred to the snow through long-wave radiation and convection currents (Federer 1968, Hillman 1971), or may decrease the snow melt rate through shading and restriction of wind movement (Federer 1968). The amount and pattern of increased snow accumulation and melt rate are strongly influenced by the size and shape of the opening relative to the prevailing wind (Gary 1974, 1975, 1979), as well as the aspect (Clausen and Mace 1972), and are variable (Clausen and Mace 1972, Gary 1979). Gary (1974, 1975, 1979) noted greater snow accumulations on the windward side and center of lodgepole pine (Pinus contorta Dougl.) clearcut strips in Wyoming and Colorado with a tapering off on the leeward side and adjacent uncut forest. Clausen and Mace (1972) found greater snow accumulations and faster melt rates on north-south oriented, 15 m wide clearcut strips of red pine (Pinus resinosa Ait.) than on east-west oriented strips in Minnesota.

The pattern and amount of snow accumulation and melt, apart from the hydrological aspects, may have important implications for wildlife. The usage of the riparian zone by wildlife should be a consideration in the design and planning of any openings to be created by forest harvesting in the riparian zone. Such openings, with the attendant changes in snow distribution, may enhance or impede wildlife

travel, foraging, etc. depending on the numerous, previously mentioned factors.

SEDIMENTATION

Soil erosion is an important influence on the degree of stream sedimentation. It encompasses two distinct erosional processes; surface erosion and mass soil movement. These processes operate naturally to produce variable amounts of stream sediment (Brown and Krygier 1971), generally over a localized area (Rice et al. 1972). Some natural soil erosion also occurs as soil and organic matter particles are dissolved as water infiltrates through the soil (Patric 1977). Surface erosion involves the movement of individual soil particles, either through suspension in water or by the action of raindrops, while mass soil movement involves movements of much larger soil amounts, either through landslides or soil creep processes (Dyrness 1967a). Surface erosion is common following soil disturbance and is generally attributed to a reduction in the infiltration capacity of the soil through compaction (e.g. by machinery) and a concomitant increase in surface runoff (Dyrness 1967a, Hillman 1971, Bormann et al. 1974, Chamberlin 1982). The removal of litter and vegetation which normally functions to retard downslope soil movement also accelerates surface erosion (Rice et al. 1972), as well as exposes soil to raindrop compaction (Rice et al. 1972, Bormann et al. 1974). The disruption of normal drainage patterns also alters the amount of surface erosion (Rice et al. 1972). Surface erosion is more closely related to watershed disturbance

factors than mass soil movement processes (Rice et al. 1972). Landslides and soil creep are more common in mountainous areas having slopes of 60 or 70% (Dyrness 1967a, Fredriksen 1970, Megahan 1972, Rice et al. 1972) and occur when gravitational and other stresses exceed the internal strength of the soil (Rice et al. 1972). Soils generally retain high water infiltration rates (Brown 1972). Human disturbance can alter the strength of the stresses on the soil, and thus the degree of mass soil movement, but usually cannot be considered as the sole factor of any movement process (Rice et al. 1972). Surface erosion normally produces increases in stream sediment levels immediately after the disturbance, followed by a gradual decline, while increased sedimentation due to mass soil movement may not be evident until several years after the disturbance (Beschta 1978).

The extent of soil erosion, particularly mass soil movement, is related to the type, stability and chemical properties of the soil (Striffler 1965, Dyrness 1967a, Fredriksen 1970, Rice et al. 1972, Swanston 1974, Chamberlin 1982), geology of the parent material (Striffler 1965, Allen and Welch 1967, Rice et al. 1972), soil water content (Rice et al. 1972, Rice 1977), climate (Rice et al. 1972), slope characteristics (Rice et al. 1972), vegetative cover (Dyrness 1967a, Bormann et al. 1974), amount of root decay (Rice et al. 1972, Rice 1977) and extent and design of forest roads (Fredriksen 1970, Megahan 1972, Rice et al.

1972, Rice 1977). Roads can affect the erosional process in a number of ways, including the gouging of subsoil, removal of downslope support, deposition of fill material on already weak slopes, interception of subsurface water and channelling of water flows. Poorly designed roads, landings and stream crossings have been reported to contribute large amounts of sediment to streams (Van Groenewoud 1977). An extensive review of sedimentation in forested watersheds is given by Swanson et al. (1982a).

Many studies, utilizing a variety of cutting patterns as well as different harvesting and extraction processes, have been carried out to determine the effects of cutting watersheds on stream sediment yields (Reinhart and Eschner 1962, Rice and Wallis 1962, Reinhart 1964, Sheridan and McNeil 1968, Fredriksen 1970, Leaf 1970, Brown and Krygier 1971, Lynch et al. 1972, Harris 1973, Bormann et al. 1974, Rice et al. 1979 are a few). In the majority of these studies little or no allowance was given for maintenance of a vegetative buffer strip along the stream banks. As a result, stream sediment yields were consistently shown to have increased, to varying degrees, following the cutting operations. The cutting of trees itself was not considered responsible for the increased stream sediment as disturbance of the forest floor was generally light and infiltration rates remained high. Rather, the extraction process (including skidtrails, landings and roads) was felt to be

responsible for the increased sediment yields, largely due to the removal of litter and subsequent mineral soil exposure and surface soil compaction, as well as interception and alteration of subsurface and surface water flows. This resulted in increased surface erosion (runoff) and/or increased mass soil movement. Because of the relatively uneven terrain of forested watersheds, water flows tend to concentrate quickly (Rice et al. 1972). The effects of increased stream sedimentation usually occurred immediately after logging and/or road construction (Reinhart and Eschner 1962, Brown and Krygier 1971, Lynch et al. 1972).

A general consensus derived from the above mentioned studies and other researchers (Chapman 1962, Dyrness 1967a, Packer 1967a, Jeffrey 1968a, 1968c, Rothacher 1970b, Lull and Reinhart 1972, Rice et al. 1972, Patric 1976, 1977, DePape and Collins 1977, Lynch et al. 1977, Patric and Kidd 1981, Chamberlin 1982) is that stream sediment yields can be considerably reduced if roads, skidtrails and landings are properly located and designed, and if the harvesting technique is designed to suit the particular area (terrain, slope, etc.). Rice (1977), for example, considered clearcutting to be much more hazardous in promoting landslides than shelterwood and selection systems (due to the greater amount of vegetation removal and subsequent decay of the root system) but cautioned that if cable yarding was to be used in the clearcutting while tractors were to be used in

the latter systems the differential in landslide risk would be considerably reduced. Soil disturbance may also be a desired management technique for promoting regeneration (Rice et al. 1972) and must be weighed against the risk of erosion. A more complete description of comparative logging methods is given in Chapter 6.

Patric (1977) felt that the discontinuity of skid-trails (surrounded by undisturbed forest floor) would lessen the impact of the erosional process as the undisturbed floor would act as a filter. He also indicated that a 30 m wide buffer strip should be left between landings and streams and that logging operations should avoid streams. Leaf (1970) left a 27 m buffer strip between a harvested area and a stream in Colorado to ensure minimum stream sedimentation. Haupt and Kidd (1965) advocated a buffer strip width of 9 m. Streeby (1971) and Van Groenewoud (1977) also noted the value of buffer strips in reducing stream sedimentation. The width of a strip necessary to minimize stream sedimentation will obviously depend on the slope of the watershed, with a wider strip necessary on steeper slopes, as well as the vegetation present and the amount of sediment expected (degree of disturbance, climate, soil, etc.) (Van Groenewoud 1977). Van Groenewoud (1977) recommended a 15 m wide buffer strip on flat terrain, and a 60 m wide strip on moderate slopes, to protect New Brunswick streams against overland

flows of sediment but cautioned that local conditions might necessitate modification.

Patric (1976, 1977) and Bormann et al. (1974) indicated that forest ecosystems in eastern North America appeared to have relatively low erosion rates. However, their emphasis was on hardwood ecosystems; it is unknown if watersheds predominated by softwoods would have comparable rates. Chamberlin (1982) felt that there were areas in western North America, at least, where potential slope stability problems should prevent cutting operations from taking place. Verry (1972) did not consider stream sedimentation to be a major consideration in clearcutting aspen on the upland portion of a watershed in Minnesota due to the low relief of the area and the rapid revegetation expected.

Regeneration has been reported to restore stream sediment levels to near normal conditions within 2 to 6 years following disturbance (Reinhart and Eschner 1962, Sheridan and McNeil 1968, Brown and Krygier 1971, Lynch et al. 1972, Rice et al. 1972, Patric 1976, 1977). However, the rate of revegetation is highly dependent on climate and soil quality (Packer 1967a) and is consequently variable. Whether new pioneer growth can restrict the erosional process with the same effectiveness as the deeper-rooted former vegetation, particularly on steeper slopes having landslide potential, will be dependent on the particular area. Tree roots rapidly lose their capacity for restricting soil

movement after the death of the parent tree (O'Loughlin 1974). Bormann et al. (1974) reported that the amount of particulate matter output was initially increased only slightly but continually increased until the fourth year following clearcutting (but no wood extraction) on a New Hampshire watershed and subsequent herbicide treatment for three summers to inhibit new growth. They concluded that this pattern was due to the ability of the relatively undisturbed (no soil compaction or litter removal) system to resist export of material but this ability declined with elimination of the new growth. Decay of the previous root network, without renewal, undoubtedly contributed to this increased output over time. The proportion of inorganic to organic matter also increased over time.

Erosion does not necessarily result in increased stream sedimentation (Packer 1967a). It may simply result in a redistribution of particulate matter which may affect future productivity, particularly on shallow soils of steep slopes or on compacted soils with reduced infiltration (Rice et al. 1972, Bormann et al. 1974).

Patric (1977) felt that disturbed soil areas such as skidtrails would be quickly restored in eastern North America due to autumn leaf fall and loosening of compacted soil by freezing and thawing. Harr et al. (1979) indicated that the adverse effects created by roads would be longer lasting than those created by logging disturbance alone.

The duration of these effects is obviously variable, dependent largely on the degree of disturbance. It has been suggested that streams having relatively stable flows (low flood flows and thus minimal flushing of streambed) will retain the effects of sedimentation longer (Sheridan and McNeil 1968). Increases in stream sediment concentrations may be only temporary (Saunders and Smith 1965, Sheridan and McNeil 1968) and, as previously mentioned, are usually most pronounced during or immediately after disturbance (Reinhart and Eschner 1962).

The burning of slash after completion of the logging operation has also been reported to increase stream sediment yields due to mineral soil exposure and its subsequent compaction by raindrop impact (Dyrness 1967a, Fredriksen 1970, Rothacher 1970b, Brown and Krygier 1971, Lull and Reinhart 1972, Rice et al. 1972). Slash may also retard downslope movement of soil (Rothacher 1970b). Rice (1977) found that fire initially reduced the risk of landslide due to reduced infiltration but this soon reversed as the roots of the former vegetation decayed and the new growth began to increase the infiltration capacity of the soil. Dyrness (1967a) emphasized that increased erosion can be avoided if the fire is of small intensity so that the litter is only charred and not completely removed. Burning of slash close to streams may also result in dramatic water temperature increases (Hall and Lantz 1969).

Another source of stream sedimentation, quite apart from the erosional sources previously discussed, is stream channel scouring through the increased stream flow which normally follows timber removal (Chapman 1962, Packer 1967a, Lull and Reinhart 1972, Aubertin and Patric 1974, Bormann et al. 1974, Patric 1976, Chamberlin 1982). In northern climates floating ice may be an important factor in this scouring effect (Patric 1976). Elson et al. (1972) concluded that the general widening (and resultant reduced water depth) of the Northwest Miramichi River in New Brunswick was due to the increased runoff produced by forest harvesting, resulting in the scouring of banks by both water and ice. Scouring would probably be particularly evident following removal of riparian vegetation and its bank-stabilizing root structure (Campbell 1970). These roots not only retard bank erosion but also greatly influence the width and depth of stream channels and the degree of bank undercutting (White and Brynildson 1967, Meehan et al. 1977, Keller and Swanson 1979).

NUTRIENTS

Increases in the concentrations of nutrients in stream water have been noted following the harvesting of watersheds (Chapman 1962, Likens et al. 1970, Rothacher 1970b, Fredriksen 1971, Brown 1972, Brown et al. 1973, Aubertin and Patric 1974). This has generally been attributed to the abrupt cessation of nutrient uptake by the harvested trees, increase in forest litter due to the discarding of unmerchantable portions of harvested trees, increase in the amount of water passing through and/or over the soil (due to reduced evapotranspiration) and more favorable conditions for breakdown of forest litter (Bormann et al. 1968, Fredriksen 1971). Likens et al. (1970) reported greatly increased exports of dissolved nutrients, particularly nitrate, and greater water acidification following deforestation (and herbicidal applicaton for the following three summers) of a hardwood watershed in New Hampshire. The loss of nitrate was assumed to be due to increased microbial nitrification (Likens et al. 1969, 1970, Likens and Bormann 1975). Brown (1972) reported nutrient losses following deforestation in Oregon but found these losses to be lower in magnitude and of shorter duration than in New Hampshire. This was attributed to the fact that the rapid revegetation was not inhibited and to the presence of heavier soils. Brown (1972) also indicated that any stream water increase in the concentration of nitrogen could have

been partially a result of the rapid invasion of the riparian zone by red alder (Alnus rubra Bong.), a prolific nitrogen fixer. The high nitrogen content of the litter of this species, and the resultant fast decomposition rate, are responsible for increased amounts of nitrogen in the soil and water (Swanson et al. 1982b). In New Brunswick the common occurrence of speckled alder (Alnus incana (L.) Moench) along stream banks may also result in substantial contributions of nitrogen to undisturbed waters. A review of the factors influencing nitrate concentration in waters draining disturbed watersheds is given by Vitousek and Melillo (1979).

Verry (1972) found no changes in the nutrient concentration of stream water one year following aspen (Populus spp.) clearcutting in northern Minnesota and attributed this to slow rates of decomposition, vigorous regeneration, low relief (little erosion) and soil characteristics. Marks and Bormann (1972) also reported that the rapid establishment and growth of pioneer species [e.g. pin cherry (Prunus pensylvanica L.f.)] following a disturbance would reduce nutrient loss.

Slash burning should be avoided to minimize nitrogen loss (Chapman 1962, Rothacher 1970b, Fredriksen 1971). Not only does burning speed up decomposition but it also eliminates the residual vegetation following the cutting operation which would have withheld some nitrogen.

Fredriksen (1971) also felt that a substantial proportion of the nitrogen was lost from a watershed in combination with sediment and that any minimization of soil loss would also minimize nitrogen loss. This would also seem to be the case with phosphorus. The containment of phosphorus by forest soils and bottom sediment is well known (Rigler 1964, M.W. Smith 1969, Peterson et al. 1973, Likens and Bormann 1975) and increases in the phosphorus content of stream water appear to be associated with increases in both organic and inorganic particulate matter (Bormann et al. 1974, Likens and Bormann 1975).

DEBRIS

Organic debris occurs naturally in all streams and, unless it is excessive, contributes to stream productivity by providing habitat diversity and channel stability (Swanson et al. 1976, Swanson and Lienkaemper 1978, Keller and Swanson 1979, Chamberlin 1982, Everest and Harr 1982). By causing pool formation and/or a slowing of stream flow, debris may reduce erosion of the streambed and banks. Organisms utilize the habitat created by organic debris, as well as the debris itself, as a substrate. This is probably more important in streams having few boulders (Murphy and Hall 1981). The movement of floating or suspended "usable" organic and inorganic matter is reduced, with the subsequent settling of some of this material on the streambed.

Many factors influence the amount of organic debris in a stream, including the age and condition of the stand adjacent to the stream, stability and steepness of the banks and slopes, ability of the stream to transport debris and rates of decomposition and physical breakdown of debris, as well as the frequency of debris torrents (Swanson et al. 1976, Swanson and Lienkaemper 1978).

The negative effects of organic debris are generally seen in periods of high flow when logs carried by the water gouge stream banks (Rothacher 1970b) and when log-debris jams force the stream to undermine and overflow its banks, causing sedimentation (Rothacher 1970b, Hillman 1971,

Narver 1971, Dominy 1972, Swanson et al. 1976, Swanson and Lienkaemper 1978). Riparian vegetation may also be destroyed (Swanson and Lienkaemper 1978). Shifting of log-debris jams will cause changes in the pool and riffle structure of streams (Meehan et al. 1969). These effects are generally less pronounced on larger rivers (Swanson et al. 1976). Additional negative aspects of excessive amounts of organic debris in streams are the prevention or delay of fish passage (Chapman 1962, Narver 1971, Burns 1972, Dominy 1972, Chamberlin 1982), increased oxygen demand (Van Groenewoud 1977) and reduced oxygen concentration near the stream bottom behind dams due to accumulation of silt and other material, resulting in less water interchange (Chapman 1962, Chamberlin 1982). These effects on oxygen concentration will be elaborated on later.

Rothacher (1970b) advocated that a buffer strip be left around streams to prevent the entry of excessive amounts of organic debris and that trees be felled and yarded away from streams. Although storms severe enough to cause any form of "debris avalanching" are rare in eastern North America (Patric 1976), logging on or near stream banks would likely contribute substantial amounts of debris to streams and provision for a buffer strip could be very worthwhile. Additionally, removal of buffer strips would also remove the natural source of organic debris input to streams which could eventually reduce stream diversity and

productivity (Swanson and Lienkaemper 1978). Chamberlin (1982) has even suggested that a reduction in the size of this debris, through the conversion of large-diameter mature forests to small-diameter pioneer growth, may also adversely affect stream productivity and stability. Van Groenewoud (1977) has indicated that the 15 m wide buffer strip recommended for New Brunswick streams for protective shading would also adequately protect most streams against excessive input of debris.

OXYGEN

The harvesting of watersheds, particularly riparian strips, influences the dissolved oxygen content of stream water in a number of ways. In undisturbed streams the oxygen content is usually at or near saturation levels (Likens et al. 1970, Rothacher 1970b). Characteristics of the stream channel such as slope, roughness (e.g. boulders) and depth influence the rate of exchange of oxygen between the water and the air (Brown 1972). Harvesting of a watershed, by producing an increase in the water yield, may increase the degree of turbulence of the water and hence the dissolved oxygen concentration (Likens et al. 1970). Conversely, any restriction of flow (as by organic debris dams), particularly during periods of low flow levels, would reduce the degree of aeration (Likens et al. 1970, Brown 1972). The decomposition of organic material (logs, slash, sawdust, etc.) by microorganisms also exerts a considerable demand on the dissolved oxygen of streams (Hall and Lantz 1969, Likens et al. 1970, Rothacher 1970b, Narver 1971, Brown 1972, Burns 1972, Dominy 1972, Van Groenewoud 1977, Chamberlin 1982). The concentration of dissolved oxygen is inversely related to the water temperature (Brown and Krygier 1967, Likens et al. 1970, Rothacher 1970b, Narver 1971, Brown 1972, Van Groenewoud 1977); thus, any water temperature increase as a result of increased stream exposure to solar radiation (due

to the removal of riparian vegetation) or ponding (caused by organic debris) will result in lower dissolved oxygen levels.

Increased water temperatures and light exposure in combination with increased quantities of nutrients and/or organic materials may result in excessive algae production (Chapman 1962, P.B. Rowe 1963, Campbell 1970, Likens et al. 1970, Narver 1971, Dominy 1972, Hansmann and Phinney 1973). In the event that the concentration of a previously limiting nutrient is increased, algae growth may be very pronounced (Chamberlin 1982). Growth of the algal biomass will initially increase oxygen levels through the photosynthetic process but, once production declines and the algae begin to decompose, a great oxygen deficit will be created (Van Groenewoud 1977). Respiration of this algae at night will also add to the oxygen deficit (Narver 1971).

CHAPTER 3

EFFECTS OF THE HARVESTING OF TIMBER ON AQUATIC LIFE

INTRODUCTION

The following chapter discusses the effects of changes in specific stream water quality parameters on aquatic life. It is not intended to be an exhaustive review of individual cases or species but rather a summary of available knowledge illustrating the numerous outcomes, as well as the degree of interaction of both variables and effects, associated with the harvesting of watersheds and, more particularly, buffer strips. Readers are invited to refer to the references presented for a more detailed examination of any particular study and/or species. Although the emphasis has been placed on fish, the importance of changes in water quality on aquatic plant and invertebrate life should not be discounted. These organisms not only serve as a food base for fish but the diversity and structure of their communities are a reflection of water quality (Wilhm and Dorris 1968, Wilhm 1970, Gaufin 1973, Patrick 1973, Hilsenhoff 1977).

The major fish species that would be adversely affected through improper logging activity in New Brunswick are salmonids and so the emphasis has been placed on this group. The term "salmonids" includes Atlantic salmon (Salmo salar L.) and several species of trout. Throughout the

text, reference is made to "less desirable, warm-water fish"; these species are generally of less value than salmonid species to fishermen and include such species as suckers (Catostomus spp.), chub (Semotilus atromaculatus (Mitchill)), yellow perch (Perca flavescens (Mitchill)) and chain pickerel (Esox niger Lesueur).

It appears, from the preceding chapter, that the majority of the adverse effects (in terms of reduced water quality) associated with the harvesting of riparian vegetation are relatively short-lived; however, individual circumstance may dictate otherwise. Any adverse changes in water quality may have an immediate negative impact on one or several age classes of fish, possibly resulting in long-lasting repercussions to the overall fish population even if water quality returns to normal within a relatively short period of time. Additionally, if an environmental change does not produce an immediate response on the part of any particular salmonid species, it may still result in a gradual species change depending on the duration (or recurrence) of the effect and its indirect influence on fish physiology, behavior, food base, etc. Salmonid species will differ in sensitivity to environmental change (Hall and Lantz 1969, Narver 1971, Burns 1972) and this sensitivity will also vary with the time of year (Lewis 1969). Fish repopulation of a stream by brook trout (Salvelinus fontinalis (Mitchill)) has been reported to occur rapidly (Phinney 1975); however,

repopulation by Atlantic salmon may take considerably longer (P. Cronin, pers. comm.).

Throughout this chapter reference is made to various stages in the life cycle of fish. The terminology used was taken from Phillips (1971). Briefly, an egg is a developing embryo inside its shell, an alevin is an embryo between the time of hatching and yolk sac absorption and a fry is a fish after yolk sac absorption through the free-swimming juvenile stage. Salmon and trout spawn in the gravel beds of streams, in areas called redds, usually emerging from these beds near the beginning of the fry stage (Phillips 1971).

The impact of any logging activity on fish populations must be judged in relation to small headwater streams. These small streams are more affected by logging disturbance than larger streams or rivers (Hall and Lantz 1969, Lantz 1971, Narver 1971, Swift and Messer 1971, Everest and Harr 1982). Headwaters provide the main spawning and rearing areas for salmon and trout (Brown and Krygier 1967, Hall and Lantz 1969, Lantz 1971, Narver 1971, Swift and Messer 1971, Everest and Harr 1982) and, as will be seen, it is during the early life cycle stages that fish are most vulnerable to adverse environment changes. The severity of changes such as reduced oxygen levels will be greatest in high density situations; thus, more productive streams may be particularly susceptible. Hanson (1973) felt that buffer strips and

other precautionary measures associated with secondary watercourses or branch streams would often be more important than those associated with the main stream.

Several general reviews on the effects of logging on salmonid populations and/or the environmental requirements of these fish have been prepared (Chapman 1962, Sheridan et al. 1967, Narver 1971, Dominy 1972, Elson et al. 1972, Elson 1974, Sharpe 1975, Lynch et al. 1977, Van Groenewoud 1977, Reiser and Bjornn 1979, Chamberlain 1982). Specific studies regarding changes in stream communities associated with logging have produced variable results, ranging from increases in stream carrying capacity (Burns 1972, Murphy and Hall 1981, Murphy et al. 1981), to little or no detectable changes (Welch et al. 1977, Englert et al. 1982), to a decrease in fish numbers (Hall and Lantz 1969, Ringler and Hall 1975). This variability is undoubtedly a reflection of the extent of disturbance (e.g. amount of residual vegetation left in riparian zone, extent of logging operation, stream crossings) and the numerous confounding factors influencing both fish numbers and the sampling technique.

TEMPERATURE

A rise in water temperature can have numerous effects on fish, including direct heat - induced mortality, increased metabolism and its associated physiological and morphological changes, increased activity of pathogenic organisms and the previously discussed reduction in solubility of dissolved oxygen (Chapman 1962, Hall and Lantz 1969, Brown 1972, Burns 1972). These effects are variable, being dependent on the time of year (and hence the stage of the life cycle) when the temperature rise occurs (Meehan et al. 1969, Lantz 1971, Ringler and Hall 1975), its duration (Hall and Lantz 1969, Lantz 1971), its magnitude, the acclimation ability of the fish species involved (Paloheimo and Dickie 1966, Lantz 1971, Lynch et al. 1977) and individual stream characteristics. Additionally, the ability of fish to acclimate to water temperature changes may decrease with increasing age of the fish (Lynch et al. 1977).

The upper lethal water temperature for salmonid fish has generally been considered to be 25 to 27°C (Fry et al. 1946, Swift and Messer 1971, Dominy 1972, Van Groenewoud 1977, Everest and Harr 1982). Javoid and Anderson (1967) reported that the preferred temperature of Atlantic salmon was 17°C while it was over 20°C for rainbow trout (Salmo gairdneri Richardson). Fry et al. (1946) established the lethal temperature limits of acclimated brook trout as 0°C and 25.3°C. Such experimental results vary, however, with

the apparatus used (Combs 1965, Javoid and Anderson 1967). Reiser and Bjornn (1979) considered an average water temperature range of 10 to 15°C to be good for anadromous salmonids while Van Groenewoud (1977) indicated normal summer highs of 13 to 20°C for trout streams in New Brunswick. Lynch et al. (1977) felt that water temperatures of 15.6 to 20°C were optimum for brook trout (a species which they considered to be very intolerant of warm temperatures) while temperatures warmer than 21.1°C would result in stress. Hall and Lantz (1969), however, observed no direct coho salmon (Oncorhynchus kisutch (Walbaun)) or cutthroat trout (Salmo clarki Richardson) mortality in water with temperatures over 25°C and attributed this to the relatively short time period each day when this temperature was exceeded. Hunt (1979), as well, could find no direct brook trout mortality in water in which summer temperatures attained a maximum of 29.5°C. It may be that the speed of temperature change and degree of temperature fluctuation, as well as the duration of the maximum temperature, are more important than the actual upper temperature reached (Hall and Lantz 1969, Van Groenewoud 1977). Direct fish mortality may also occur as nitrogen gas comes out of solution, due to the reduced solubility in warmer water, resulting in bubbles which can destroy tissue or clog the bloodstream (Lantz 1971).

The indirect effects of increased water temperature may have a much greater impact on fish, particularly

salmonid, survival. An increase in temperature will be accompanied by an increase in metabolic activity with the result that more food energy has to be utilized in maintenance functions rather than growth (Baldwin 1956, Beamish 1964a, Lantz 1971, Dominy 1972). If this expenditure of metabolic energy is excessive, particularly in relation to the available food, normal processes such as swimming ability will be adversely affected (Lantz 1971, Swift and Messer 1971, Reiser and Bjornn 1979). The severity of this will vary by season and the associated metabolic requirements (Lantz 1971). Weight loss and starvation may occur due to a lack of food or a reduction in either feeding ability or assimilation of food (Latta 1969). Additionally, less dissolved oxygen will be available to meet the requirements of increased respiration (Lantz 1971, Davis 1975).

Salmon and trout prefer colder water temperatures, and associated higher dissolved oxygen levels, than most other fish species (Lantz 1971, Dominy 1972, Van Groenewoud 1977). Competition from the less desirable, warm-water fish will increase as water temperature rises (Chapman 1962, Lantz 1971, Brown 1972).

The normal behavior patterns of spawning, egg incubation, fry emergence and migration may be altered by temperature changes due to the effects of temperature on physiology (Henderson 1963, Meehan et al. 1969, Bjornn 1971, Lantz 1971). The input of cooler ground water may cause

brook trout to congregate along the edge of a stream (Latta 1969). Ringler and Hall (1975) speculated that any temperature increase which results in earlier fry emergence, through acceleration of the development rate, may affect the future survival of the fry. Lantz (1971) suggested a relationship between the temperature at which adult fish spawn and the survival of their young. Temperature and light changes may also alter the invertebrate and primary producer community composition and behavior (Chapman 1962, Holt and Waters 1967, Waters 1969, Hansmann and Phinney 1973, Van Groenewoud 1977, Fraley 1979) which will necessitate a change in fish feeding behavior if fish numbers are to remain stable. Alteration of light periods has been shown to affect the drift of aquatic invertebrates (Holt and Waters 1967); changes in water temperature and amount of stream flow may also influence the amount and timing of this drift.

Higher water temperatures often result in an increase of algae, parasites, pathogenic bacteria, diseases and viruses (Brown and Krygier 1967, Sadler 1970, Lantz 1971). The resistance of salmonid fish to these agents is usually reduced due to the changes in metabolic activity. Dissolved chemicals, such as those leached from organic debris in the water, would also provide additional stress at higher water temperatures (Lantz 1971). Algal blooms may reduce the flow of intergravel water by filling up the

interstitial spaces in the gravel (Chamberlin 1982); the significance of this will be discussed later.

Conservative increases in stream water temperature may actually be beneficial to salmonid fish in cold, low-productivity streams through an increase in the production of algae, bacteria, insects, etc., which would provide sufficient food source to accommodate the increased metabolic activity of fish, resulting in increased fish growth (Sharpe 1975, Murphy and Hall 1981, Murphy et al. 1981). Coincident increases in stream water nutrients may also have a similar effect (Lynch et al. 1977). Burns (1972) reported examples from California where slightly increased stream temperatures increased salmonid biomass. Englert et al. (1982) noted a trend of increased fork length and/or weight of Atlantic salmon parr as the amount of canopy cover was reduced along streams in New Brunswick and Nova Scotia. Chapman and Knudsen (1980) concluded that light was an important limiting factor to salmonid production in Washington streams and that prudent removal of stream side vegetation along certain stream sections, to limit water temperature increases and minimize sedimentation, might result in an increase in biomass. Any increase in invertebrate biomass due to increased stream exposure may also be accompanied by a shift in species since those species inhabiting riffle areas would be less affected by increased amounts of stream sediment than those inhabiting pools (Murphy and Hall 1981). Newbold et

al. (1980) found higher densities of some macroinvertebrate species (generally those with short generation times) along sections of California streams without buffer strips as compared to sections with buffer strips; overall macroinvertebrate diversity was, however, higher in streams having buffer strips.

The above discussion regarding the effects of temperature change have dealt primarily with water temperature increases, rather than decreases. As previously discussed, increased summer water temperatures commonly follow riparian vegetation removal but the effect on winter water temperature has not been as clear. In the event that water temperatures are lowered, changes in fish physiology and community composition would also be expected. Lantz (1971) has indicated that a lowering of water temperature may have a more adverse effect on salmonid fish than raising it, due to the relatively long time needed to acclimate to cold temperatures. If incubation periods for fall and winter-spawning fish are increased, growth may be reduced and mortality may be high as the fish are susceptible to stream bed alterations for longer periods of time (Chapman 1962).

STREAM FLOW

Alteration of stream flow, through the removal of timber on a watershed, may be either beneficial or detrimental to aquatic life. Any increase in water flow might increase the amount of living space and/or spawning habitat available for fish and fish food (algae, insects, etc.) production, as well as providing a greater catchment for terrestrial food sources (Chapman 1962, Hall and Lantz 1969, Burns 1972, Dominy 1972, Reiser and Bjornn 1979, Chamberlin 1982). Water flow velocities and the size of riffles would be more affected than pools (Reiser and Bjornn 1979). Spawning and nursery areas, in particular, may become more accessible to adults (Narver 1971, Dominy 1972, Reiser and Bjornn 1979). Additionally, an increase in summer stream flow may partially offset any water temperature increase produced from greater stream exposure due to the larger volume and/or velocity of water flow. Current velocity itself may be an important factor influencing salmonid population density through its modification of available food (Gibson 1978, Lewis 1969).

The potential impact of increased stream flow in contributing to stream sedimentation through stream channel and bank scouring has previously been reviewed. White and Brynildson (1967) mentioned the adverse effects of stream flooding in reducing stream bank stability and fish habitat through the breakdown of overhanging banks and removal of

beneficial logs and brush which provided habitat. Increased gravel bed movement due to higher peak flows can also jar eggs and alevins and even scour them from their redds (Hall and Lantz 1969, Phillips 1971, Burns 1972). Chapman (1962) noted that abnormally high flows and associated shifting of gravel may dislodge aquatic organisms. Saunders and Smith (1965) indicated that this scouring action was also beneficial in that it removed sediment from the bottom of trout streams in Prince Edward Island. The comparatively higher gradient of most New Brunswick streams, and consequent greater degree of flushing under normal stream flow conditions, may reduce the significance of this beneficial aspect (P. Cronin, pers. comm.). Peters (1967), as well, suggested that stream velocity had an indirect influence on the amount of stream sediment deposition. Hansen (1971) attributed minimal amounts of streambed sand to the transport capacity of pool and riffle streams. Whether or not above average runoff reduces the amount of deposited sediment largely depends on how much sediment the stream was carrying when flows subsided (Bjornn et al. 1977). Although dependent upon many factors, moderate seasonal flooding may also benefit fish populations indirectly through deposition of silt in the riparian zone, thereby increasing soil productivity (Jahn 1979, Odum 1979).

It appears that a relatively stable stream flow, without excessive flooding or drought, is the most

beneficial to salmonid production (Narver 1971, Dominy 1972). Extreme water fluctuations and reduced stream flow have both been shown to be detrimental to fish populations (Corning 1970, Kraft 1972).

SEDIMENTATION

The adverse effects of excessive amounts of stream sediment on salmonid populations have been reviewed in depth by Cordone and Kelley (1961), Phillips (1971), Ritchie (1972) and Reiser and Bjornn (1979). Specific studies illustrating the detrimental effects of sediment on survival in fish and aquatic invertebrate populations are numerous (Tebo 1955, Coble 1961, Saunders and Smith 1965, Shelton and Pollock 1966, Peters 1967, Hausle and Coble 1976, Bjornn et al. 1977).

High concentrations of suspended sediment may cause adult fish mortality through clogging and physical abrasion of the gills resulting in carbon dioxide retention and reduced oxygen intake (Narver 1971, Phillips 1971, Dominy 1972, Ritchie 1972, Lynch et al. 1977, Reiser and Bjornn 1979). The quantity of sediment required to bring about this direct mortality has reportedly been variable (Phillips 1971, Ritchie 1972) due to the number of factors involved such as the species, age and general health of the fish, the size, shape and hardness of the sediment, the length of exposure and water quality (Phillips 1971). Of more importance to adult fish are the indirect effects of sediment such as an added physiological stress making them more susceptible to other mortality factors (Herbert and Merkens 1961, Phillips 1971, Ritchie 1972) and increased turbidity which may interfere with migration patterns

(Reiser and Bjornn 1979) and feeding efficiency (Narver 1971, Phillips 1971, Dominy 1972, Reiser and Bjornn 1979, Everest and Harr 1982). Young fish may benefit from this increased turbidity through a reduction in predation (Ritchie 1972); however, such a benefit would only be of a short-term nature (J. Gilbert, pers. comm.). Because salmonids are generally more dependent upon sight for feeding than the less desirable, warm-water fish (e.g. suckers) any increase in turbidity may give these latter species a competitive advantage (Phillips 1971).

The deposition of sediment on streambeds will have substantial effects, both direct and indirect, on the survival and distribution of salmonid eggs, alevins and fry, as well as aquatic invertebrates. Salmon and trout require clean, loose gravel on the streambeds for spawning (Cope 1966, Dominy 1972) and eggs are usually buried 15 to 30 cm deep in this gravel (Cope 1966). The location (usually near the tail of a pool) and shape of a redd is such that the exchange of intergravel and surface water is maximized (Phillips 1971). This ensures a continuous input of oxygen for the developing eggs and alevins, as well as removal of metabolic wastes such as carbon dioxide and ammonia. Shumway et al. (1964) indicated that embryo development may, in some cases, be inhibited not by a lack of oxygen but by the level of metabolic waste. Sedimentation reduces survival by decreasing the permeability of the gravel bed to water

and hence the velocity of intergravel flow and rate of water interchange (Coble 1961, Chapman 1962, Silver et al. 1963, Shumway et al. 1964, Shelton and Pollock 1966, Peters 1967, Hall and Lantz 1969, Narver 1971, Phillips 1971, Brown 1972, Dominy 1972, Hausle and Coble 1976, Van Groenewoud 1977). Any decrease in water flow (e.g. by organic-debris dams), or increase in debris settled on the gravel, will also reduce the exchange of surface and intergravel water and thus adversely affect egg and alevin survival (Hall and Lantz 1969, Brown 1972, Dominy 1972, Ringler and Hall 1975, Van Groenewoud 1977, Reiser and Bjornn 1979, Chamberlin 1982). In addition, deposition of sediment is enhanced in pools formed behind organic debris dams (Chapman 1962). The severity of this reduced water interchange is obviously a function of the total number and density of developing embryos (Coble 1961). It should be mentioned that the dissolved oxygen in the intergravel water of a stream is supplied in part by interchange with ground water, in addition to surface water (Vaux 1962). However, Sheridan (1962) found low dissolved oxygen concentrations in ground water. Deposited sediment may also increase the temperature gradient between surface and intergravel water due to the reduced interchange of water (Ringler and Hall 1975).

Sediment deposited on the streambed can impede the emergence of fry from the redds (Shelton and Pollock 1966, Hall and Lantz 1969, Sadler 1970, Phillips 1971, Burns 1972,

Hausle and Coble 1976, Van Groenewoud 1977). Oxidation of sediment also occurs (Hargrave 1972), contributing to lower dissolved oxygen levels.

The filling of gravel bed interstices with sediment and small debris reduces the amount of shelter for both newly emerged fry and aquatic invertebrates (McCrimmon 1954, Narver 1971, Phillips 1971, Ritchie 1972, Meehan et al. 1977, Van Groenewoud 1977). This may substantially reduce the number of young fish which a stream can support over the winter (Bjornn et al. 1977).

Aquatic primary production may be adversely affected by turbid water as the amount of light available for photosynthesis is reduced (Cordone and Kelley 1961, Chapman 1962, Narver 1971, Phillips 1971, Dominy 1972, Ritchie 1972, Lynch et al. 1977). Sediment may also physically harm algae and other forms of aquatic plant life (Cordone and Kelley 1961) and may reduce the overall food supply through a decline in the amount of detritus which is trapped on the streambed (Van Groenewoud 1977).

The minimal amount of sediment required to produce adverse effects is obviously variable depending on the duration of the sedimentation process, other stream parameters and fish density. A wide range of acceptable sediment concentrations are presented in the reviews by Herbert and Merkens (1961), Phillips (1971) and Ritchie (1972). Phillips (1971) suggested that direct fish mortality would

occur after several days at concentrations over 200 to 300 parts per million (p.p.m.) while Herbert and Merkens (1961) and Ritchie (1972) cited much higher values, although Herbert and Merkens (1961) did find concentrations as low as 270 p.p.m. to be harmful. As a general guideline for good anadromous salmonid rearing habitat, Reiser and Bjornn (1979) recommended that suspended sediment concentrations be less than 25 mg/L with riffles having less than 20% fine sediment.

DEBRIS

Several of the negative effects of logging debris in streams have previously been discussed and will not be included here.

The decomposition of organic debris in streams not only increases the oxygen demand but also releases soluble organic substances, such as wood sugars and tannins, which create additional oxygen demand and may reduce the amount of solar radiation available for photosynthesis (Narver 1971). Nutrient release from the decomposition of this debris will promote algal growth (Narver 1971). Large amounts of slime bacteria (Sphaerotilus spp.), considered detrimental to eggs and alevins, may also be produced during this decomposition process (Gordon and Martens 1969).

OXYGEN

Numerous studies have outlined the many adverse effects of lowered dissolved oxygen levels on salmonid populations, including reduced survival and emergence of embryos, reduced size of alevins, delayed hatching and reduced swimming performance, as well as other forms of lowered fitness resulting in weaker, less competitive fish (Wickett 1954, Coble 1961, Davis et al. 1963, Silver et al. 1963, Beamish 1964b, Shumway et al. 1964, Bouck and Ball 1965, Whitworth 1968, Hall and Lantz 1969, Kutty and Saunders 1973). Salmonid fish actively avoid areas of low oxygen concentration (Whitmore et al. 1960).

An extensive review on the effects of changes in oxygen concentration, as well as the oxygen requirements of fish and aquatic invertebrates, is given by Davis (1975). Hynes (1970) indicated that less than 75% oxygen saturation would reduce trout activity. Reiser and Bjornn (1979) felt that oxygen concentration should be at saturation for rearing anadromous salmonids. Whitworth (1968) found that few yearling brook trout were able to tolerate oxygen levels of 2.0 mg/L or lower and Silver et al. (1963) indicated that no steelhead trout (Salmo gairdneri Richardson) or chinook salmon (Oncorhynchus tshawytscha (Walbaun)) embryos survived at oxygen levels of 1.6 mg/L. It is important to remember that oxygen requirements vary by the species, time of year, temperature, activity and stream characteristics (e.g. rate

of flow, gravel size) (Whitmore et al. 1960, Davis 1975, Ringler and Hall 1975). Allen (1969) felt that the oxygen requirements of fish should be determined relative to their required levels of activity. The timing of any changes in dissolved oxygen levels in relation to the stage of development is a very important factor in determining the effect on fish populations. For example, Ringler and Hall (1975) reported that survival of coho salmon to emergence from the redds was relatively unaffected by reduced oxygen levels as the lowest oxygen levels occurred after the critical hatching period.

ADDITIONAL EFFECTS

The role of riparian vegetation in the prevention of water temperature increases, water sedimentation and excessive amounts of organic debris in streams has been discussed. Disturbance to the riparian zone may, however, have additional effects on the ecology of aquatic life in streams.

The stabilizing action of the roots of riparian vegetation (trees, shrubs and herbs) in the prevention of streambank erosion, and the subsequent formation of undercut banks and deeper channels, has previously been mentioned. These undercut banks, as well as overhanging riparian vegetation, are extremely important as cover for salmonid fish and are used for resting, escape, wintering and feeding (White and Brynildson 1967, Narver 1971, Dominy 1972, Meehan et al. 1977, Reiser and Bjornn 1979, Chamberlin 1982). Rainbow and brook trout appear to be mainly photonegative with a tendency to remain under bank cover (Gibson and Keenleyside 1966, McCrimmon and Kwain 1966, Kwain and MacCrimmon 1969) while young Atlantic salmon appear to be predominately photopositive (Gibson and Keenleyside 1966). Numerous studies have found a direct relationship between the amount of cover in a stream and the biomass of trout (Boussu 1954, Elser 1968, Lewis 1969, Mills 1969, O'Connor and Power 1976, Gilbert 1978). Attempts at stream habitat alteration which have included an increase in the amount of

available cover have resulted in greater numbers of salmonids, particularly trout (Saunders and Smith 1962, Hunt 1969, 1971, 1976, Hall and Baker 1982, Reeves and Roelofs 1982). Hunt (1969), in Wisconsin, concluded that higher trout biomass was a result of decreased overwinter mortality, rather than increased recruitment into the altered stream section or increased growth of individuals. Any logging operation involving tree removal on (or near) stream banks, felling of material in or yarding of material across (or through) streams has the potential to destroy salmonid cover (Chamberlin 1982).

The input of both plant and animal matter into streams from the riparian zone can be a very important source of energy for salmonid populations, particularly in small, cold-water streams where shading limits primary production (Chapman 1962, Nelson and Scott 1962, Minshall 1967, Hall and Lantz 1969, Burns 1972, Fisher and Likens 1972, Meehan et al. 1977, Van Groenewoud 1977, Reiser and Bjornn 1979, Chamberlin 1982). Riparian vegetation essentially controls the balance between autotrophy and heterotrophy in a stream by influencing the amount of light available for primary production and the amount of terrestrial nutrients reaching the stream water (Cummins 1979, Jahn 1979). This control by riparian vegetation is reduced on wider streams. Small streams are basically heterotrophic while larger ones tend to be more autotrophic up to a point where, due to

increased water depth and turbidity, heterotrophy again predominates (Meehan et al. 1977, Jahn 1979). The importance of the autotrophic process in streams may, however, be underemphasized (Minshall 1978). Sufficient dissolved oxygen levels in the small, cold-water heterotrophic streams are maintained by water turbulence (Fisher and Likens 1973). An extensive review of the types and rates of energy inputs in streams is provided by Triska et al. (1982).

Terrestrial matter which enters streams may be utilized by fish directly (e.g. feeding on terrestrial insects) or indirectly, through decomposition of the energy source by decomposer microorganisms or invertebrates. Terrestrial invertebrates which accidentally fall into streams often constitute an important part of the diet of salmonid fish and are usually most plentiful during the summer and early fall when benthic invertebrates are normally less abundant and when salmonid food consumption and growth is greatest (Hunt 1975). In addition, there is little competition for this food source from warm-water fish species as they do not readily utilize surface-drift foods (Hunt 1975). Studies regarding the cycling of nutrients in streams (Fisher and Likens 1973, Cummins 1974, 1979, Hynes et al. 1974, McDowell and Fisher 1976) have emphasized the tremendous influence of the surrounding terrestrial vegetation and the capacity of the stream to process received material. The microorganisms of small streams appear to be

very adept in processing organic matter, even in cold water temperatures (Cummins 1974, McDowell and Fisher 1976, Meehan et al. 1977). The amount of terrestrial matter actually utilized in the stream is dependent on many factors such as the amount of input and stream flow (influences not only the export of matter from the system but also oxygen levels, sediment loads, etc.). Woody debris requires a longer decomposition time than does leafy material but is usually available to the stream for a longer period of time (Meehan et al. 1977). Hardwood leaves and branches generally decompose faster than those of softwoods (Gosz et al. 1973, Meehan et al. 1977) and different hardwood and softwood species will differ in decomposition rates (Gosz et al. 1973, Hynes et al. 1974). O'Connor and Power (1976) noted lower food availability in streams flowing through coniferous forests.

Input of terrestrial matter into streams may also be detrimental to fish populations. Autumn leaf fall from the riparian zone may bring about a decrease in water quality due to the increase in dissolved solids (Rothacher 1970b). White and Brynildson (1967) suggested (in their guidelines for trout stream habitat improvement in Wisconsin) that trees should be removed from riparian zones, provided no excessive water temperature increases were expected, as they inhibited the growth of low stream side and aquatic plants. Hunt (1979) experimentally removed the

woody vegetation (primarily speckled alder) from strips along trout streams in Wisconsin to increase primary production and trout biomass, but cautioned that further testing was required to determine the most advantageous degree of removal.

Speckled alder is common in areas of shallow water tables (Healy and Gill 1974, Knighton 1981) and stands of this species appear to be long lasting (Brickman 1950). The value of alder as a nitrogen fixer has previously been mentioned. This species also retains its leaves, thus providing shade, for a relatively long period of time in the fall. Additionally, it is more important than conifers in the value of food provided to stream ecosystems (Chamberlin 1982). However, White and Brynildson (1967) felt that alder provided too much shade on small streams and that the lower branches, which could serve as cover for fish if they dropped into the water, were usually missing.

Any change in the composition of the riparian vegetation (e.g. through harvesting) may have a profound effect on this energy balance process. If the degree of stream shading by the riparian vegetation is altered the balance between autotrophy and heterotrophy may be changed; if a species change occurs in the riparian zone (e.g. pioneer growth following cutting) the decomposition process may be altered, as well as the species composition of decomposer organisms (Fisher and Likens 1973, Cummins 1974, 1979,

Meehan et al. 1977, Van Groenewoud 1977). The duration, and indeed the permanence, of any alteration will be a function of the regrowth capacity of the pre-logging vegetation, longevity of invading or pioneer vegetation and degree of site disturbance. Lack of empirical data makes it impossible to uniformly qualify the effect of this change in energy base on salmonid populations for any set of conditions.

SUMMARY

The preceding two chapters illustrate the complexity and variability encountered in attempting to establish a definitive answer as to the effect of buffer strip removal on aquatic populations or the amount of buffer strip removal which can be tolerated by aquatic populations before adverse effects are detected. The magnitude of change of any particular water quality parameter will vary with the degree of disturbance (e.g. amount and type of buffer strip removal) and the individual watershed, but must also be interpreted relative to the stage of the life cycle of aquatic life. Effects (beneficial or adverse) may have relatively little importance to aquatic life if they occur during an insensitive period of the life cycle. Additionally, effects may alleviate or enhance each other (e.g. increased stream flow may prevent excessive temperature rises or may remove excessive amounts of deposited sediment, logging debris may promote sedimentation) dependent, of course, on the temporal occurrence of these changes.

This complexity is reflected in the great deal of variability, and its subsequent effect on statistical analyses, encountered by the majority of studies investigating changes in aquatic populations produced by buffer zone alteration. The lack of uniform data has, in some cases, led to decisions which are based on reasonable speculation rather than facts. It is not intended to imply that such

decisions are erroneous but simply to emphasize the difficulty in establishing a definitive cause and effect relationship. This is evident in the consistent reference to the relatively greater importance of the indirect effects, rather than the direct effects, of changes in water quality parameters on aquatic life. The adverse effects of excessive amounts of deposited sediment appear to be fairly easily demonstrable and should be of primary concern to any manager involved with the harvesting of buffer zones. The effects of increased sedimentation may not be evident in small, high gradient streams but may be detected further downstream once current velocity is reduced (Murphy and Hall 1981, Murphy et al. 1981). Additionally, in those cases where stream productivity is increased due to the greater exposure to light, the adverse effects of sedimentation on productivity may not be evident until the stream side vegetation recovers (Murphy and Hall 1981).

It appears that the width, and indeed the value, of a buffer zone should be judged on an individual basis not only considering the stream itself but also the present vegetation, the amount of disturbance associated with any reduction in the buffer zone and the expected rate of regrowth. Lantz (1971), Marlega (1971) and Streeby (1971) have emphasized that the width of a buffer zone necessary for stream protection should be flexible. Van Groenewoud (1977) and Welch et al. (1977) concluded that buffer zones

60 m wide (which were advocated in a 1968 amendment under the New Brunswick Crown Lands Act) were generally unnecessary in New Brunswick. Narver (1973) suggested that a 9 m wide buffer zone might be sufficient for small streams in British Columbia while the United States Department of the Interior (1970) recommended a 23 m minimum width. Newbold et al. (1980) found buffer zone widths of 30 m or greater to be generally effective in preventing any logging-produced alteration of the benthic community of California streams but added that this width could be considerably reduced during careful logging operations. Aubertin and Patric (1974) reported minimal increases in stream turbidity following the clearcutting of a watershed in West Virginia in which a 10 to 20 m wide buffer zone was reserved along the entire length of the stream (selection cutting took place within the zone as well) and roads were located at least 30 m from the stream.

It may be advisable to establish "safe" guidelines, in terms of prevention of adverse effects to aquatic life under most conditions. Then, after consideration of the harvesting operation and other local conditions, these "safe" limits may be modified. This appears to be the recommendation of the New Brunswick Department of Natural Resources (1981). Minimum buffer zone widths of 15 m, 30 m and 60 m were advocated for streams which were 0.5 m or wider (during normal summer flow) on slopes of 0 to 5%, 5 to

10%, and greater than 10%, respectively. However, harvesting could be carried out within the zone if so authorized by the Regional Resource Manager. The further suggestion that large streams, rivers and lakes may require wider buffer zones is, apart from the standpoint of aesthetics, less understandable as these larger water bodies are generally more resistant to logging - produced changes in water quality parameters. As well, it is in the smaller streams that the most susceptible early stages in the life cycles of salmonids normally occur. However, adverse conditions in large water bodies, once created, may be relatively long lasting (Anonymous 1982a).

The extent of any allowable buffer zone reduction should depend not only on the many previously discussed factors (slope, soil characteristics, stream orientation, etc.) but also largely on the care and planning associated with the road network and the logging and extraction processes. This will be elaborated on in a later chapter but a few points can be mentioned now. Stream crossings by roads must be properly constructed and kept to a minimum. At no time should logs be skidded across or through streams, nor should equipment enter or cross through streams. Any such occurrences may result in persistent sediment sources. Stream crossings associated with logging operations have been shown to have a negative effect on salmonid populations in New Brunswick and Nova Scotia (Englert et al. 1982). To

preserve the integrity of the stream channel and banks it has been recommended that no heavy equipment be allowed to work within 10 to 15 m of any stream (Van Groenewoud 1977, New Brunswick Department of Natural Resources 1981). Adequate care should also be taken near any identified small, intermittent streams or springs supplying substantial inputs of water to streams which are utilized for spawning.

Adequate shading for small streams may be provided by unmerchantable tree species such as alder and willow (Salix spp.), or low stream side vegetation such as shrubs and grasses. In these areas removal of all merchantable trees may be tolerable (at least from the standpoint of fish populations) providing measures to prevent stream sedimentation and logging debris in streams are taken. Selection cutting may be the most advisable in other areas in order to preserve the beneficial function of the buffer zone. In some cases adverse conditions, such as steep slopes or unstable soils, should preclude any reduction of the buffer zone.

CHAPTER 4

EFFECTS OF THE HARVESTING OF TIMBER ON WILDLIFE

The impact of any buffer zone harvesting operation on wildlife will be governed to a larger degree by the amount of disturbance to the surrounding stands than was the case with aquatic populations. As a result definitive guidelines will be even harder to establish. This discussion will attempt to illustrate the value of the riparian zone to wildlife species in general. Based on this brief overview it may be possible to deduce the importance of any particular watershed riparian zone to wildlife species in an area, as well as enabling a resource manager to be aware of and to evaluate any potential problems arising from a proposed harvesting operation.

Disproportionately heavy usage of riparian zones by wildlife, reflected in both the diversity and number of avian and mammal populations, has been well documented (Carothers and Johnson 1975, Curtis and Ripley 1975, Kelly et al. 1975, Miller et al. 1976, Hubbard 1977, Stevens et al. 1977, Best et al. 1979, Ewel 1979, Hehnke and Stone 1979, Jahn 1979, Odum 1979, Thomas et al. 1979a). Animals that feed and/or reproduce in or near water are very dependent upon the riparian zone (Thomas et al. 1979a). These include various reptiles and several well known mammals such as beaver (Castor canadensis Kuhl), muskrat (Ondatra

zibethicus (L.)), mink (Mustela vison Schreber), otter (Lontra canadensis (Schreber)) and water shrew (Sorex palustris Richardson). Riparian zones are popular nesting, resting and feeding sites for many species of birds (Carothers and Johnson 1975, Curtis and Ripley 1975).

The attractiveness of riparian zones to wildlife is undoubtedly related to the structural and species diversity of the vegetation (and subsequent "edge effect") found in these zones, due largely to the high degree of variation in factors affecting plant growth (such as water table levels, degree of flooding, seedbed conditions, nutrient and light availability, slope) over relatively short distances from the waters edge (Curtis and Ripley 1975, Kelly et al. 1975, Hubbard 1977, Campbell and Franklin 1979, Ewel 1979, Jahn 1979, Odum 1979, Thomas et al. 1979a). Availability of light and lack of competition from tree species on or near stream banks provide favorable conditions for many species of shrubs, forbs and grasses. Periodic flooding (with attendant deposition of sediment), availability of moisture (thus preventing drought conditions) and favorable microclimate (Swanson et al. 1982b) in riparian zones enhances the growth and productivity of many vegetative species (Carothers and Johnson 1975, Jahn 1979, Odum 1979, Thomas et al. 1979a, Boer et al. 1982, Swanson et al. 1982b). This enhanced growth may provide additional wildlife habitat and/or food.

Conifer tree species are a common occurrence along the banks of waterways in New Brunswick. Moen (1968), Ozoga and Gysel (1972) and Drolet (1976) have reviewed the protective advantages of dense conifer forests to white-tailed deer (Odocoileus virginianus (Zimm.)) in northern climates, specifically the reduction in snow depth (due to the interception of snow by the canopy), reduced wind velocity and beneficial radiation aspects (which result in warmer winter air temperatures and conservation of body heat). Riparian zones generally provide some degree of topographic protection (Boer et al. 1982) as well as a humid, shaded atmosphere for wildlife (Thomas et al. 1979a). The common association of deer wintering areas with waterways in New Brunswick (A. Boer, pers. comm.) emphasizes the importance and sensitivity of the riparian zone. In addition, undisturbed riparian zones may suffice as wintering areas for small numbers of deer (Boer et al. 1982).

Riparian zones have been utilized as transportation corridors for a variety of wildlife species (Sparrowe and Springer 1970, Zagata and Haugen 1974, Kelly et al. 1975, Thomas et al. 1979a). Undisturbed buffer zones undoubtedly function as shelter for species feeding in recently cut upland sites as well. Openings created within a buffer zone may reduce its effectiveness as a transportation corridor due to greater snow depths which may impede animal movement, particularly ungulates (as mentioned in

Chapter 2). Animals formerly utilizing the zone as a corridor may also actively avoid crossing open areas. For example, marten (Martes americana (Turton)) have demonstrated an avoidance of areas not having overhead cover (Hawley and Newby 1957).

Thomas et al. (1979b) cited studies regarding the value of available water sources to deer and elk (Cervus elaphus L.). However, Mackie (1970) found the distribution of water sources to have little influence on the distribution of either mule deer (Odocoileus hemionus (Rafinesque)) or elk in Montana and Moen (1973) concluded that wild ruminants were not entirely dependent upon open water sources if snow or succulent vegetation were available. Moen (1981), however, did indicate that white-tailed deer would use available open water. Hoskinson and Mech (1976) reported that the majority of the summer ranges of radio-tagged white-tailed deer in Minnesota were associated with major waterways and postulated that water was used to escape predation by wolves (Canis lupus L.). In New Brunswick, summer ranges of white-tailed deer have also been found to be oriented parallel to waterways (Boer 1978).

The impact of any particular buffer zone harvesting operation on wildlife will primarily depend on the amount of prior usage of that zone by various wildlife species and the proximity and accessibility of other undisturbed buffer zones to these species. The condition of the

surrounding stands and extent of disturbed area are also very important in assessing this impact, particularly for species requiring a transportation corridor or undisturbed, mature forest. Species which require mature forest such as the ovenbird (Seiurus aurocapillus (L.)) (Best et al. 1979), cavity-nesting birds (Myers and Morris 1975) and marten (Dodds and Martell 1971, Banfield 1974) would be adversely affected by harvesting in the buffer zone. Other species such as the song sparrow (Melospiza melodia (Wilson)) (Best et al. 1979) may benefit from such disturbance. Myers and Morris (1975) have discussed the potential effects of various watershed management practices on bird species and concluded that the increased habitat diversity produced by patch or strip cutting and thinning would benefit most, but not all, species. Most mammal species would also profit from this increased habitat diversity. Beaver may benefit from openings created in the canopy near streams provided young aspen growth invades these openings (D. Cartwright, pers. comm.).

Stream bank disturbance due to harvesting in the riparian zone may destroy the burrows and dens of mammals such as mink and muskrat. Additionally, adverse effects on fish populations may indirectly affect species such as mink which utilize fish as a food source. Suspended sediment may also directly affect aquatic mammals. Alteration of the

terrestrial invertebrate community through changes in the vegetation may affect bird species as well.

A concluding statement regarding the overall effects on wildlife species of harvesting timber in a riparian zone would be very similar to that given in the previous chapter, dealing with effects on fish populations. The characteristics and value to wildlife of each riparian zone are different (Thomas et al. 1979a). It should be the responsibility of the resource managers to identify the value of each riparian zone on an individual basis and, based on the expected impact associated with harvesting, make a decision as to the type and amount of timber removal, if any. Although the values of riparian zones to wildlife species are generally harder to define than the values to fish species, this should not be interpreted to mean that riparian zones are of less importance to wildlife. Indeed the opposite may be true, particularly if the zone provides one or more limiting factors which are critical to a local population (e.g. undisturbed conifer cover utilized as a deer wintering area and surrounded by heavily cut areas). As mentioned in the preceding chapter, some form of selection cutting may be a less extreme modification of the riparian zone than clearing patches or strips and may represent a reasonable alternative in some cases.

CHAPTER 5

BUFFER ZONE CONDITION

The preceding two chapters have dealt with considerations of fish and wildlife which any resource manager should incorporate into buffer zone management. A third very important consideration is the present condition of the buffer zone and the anticipated duration of its effectiveness in providing benefits to fish and wildlife species. All living plant communities are subject to change through the natural processes of death, decay and regrowth, and buffer zones are no exception.

The optimum benefits of any buffer zone to fish and wildlife populations will exist only as long as the zone perpetuates itself, which is a function of many factors such as the plant species present, soil types, regeneration and overall site quality. Gradual or localized changes within the buffer zone will normally have little affect on the benefits derived from the zone by fish and wildlife; widespread changes such as extensive windfall or tree defoliation may severely reduce these benefits.

The beneficial aspects of any buffer zone will vary between the different plant species present (Streeby 1971). A change in the vegetative species composition within the buffer zone, either naturally or through human disturbance (e.g. logging), will produce an alteration in

the benefits provided by any particular zone to aquatic life and wildlife. This report was not intended to be an extensive silvicultural review nor a review of the effects of various harvesting systems and silvicultural treatments (e.g. degree of mineral soil exposure) on the propagation of different plant species. Numerous sources of information are available on this subject. Moore and Boer (1977) provided such a literature review related to deer wintering area management in New Brunswick which would also be generally applicable to riparian zones.

Decisions regarding wood extraction in buffer zones are obviously tempered to a large degree by the value of the wood in relation to the additional costs associated with extra care and planning of the logging operation. The amount of residual merchantable trees and unmerchantable vegetation (trees, shrubs, grasses) retained in the harvested buffer zone will have an important influence on the effects of the cutting operation on fish and wildlife populations. The potential effect of partial timber removal on the residual vegetation must be carefully evaluated. For example, the degree of windfall following selection or partial cutting, clearcutting of adjacent areas or road construction will depend on the species involved, age and condition of the vegetation, local site conditions (e.g. soil, water) and degree of wind protection (e.g. by topography).

The resource managers must therefore be able to assess the condition of any particular riparian zone in terms of its present and future economic value as well as the benefits it confers on fish and wildlife species. With proper planning and harvesting it may be possible to accommodate all concerns and even increase the future benefits through enhancement of the functional longevity or quality of the buffer zone.

CHAPTER 6

LOGGING METHODS

The importance of careful logging and choice of the proper logging methods to minimize soil disturbance, as well as the proper location and design of roads and stream crossings, should be obvious in light of the previous discussion, particularly with regard to stream sedimentation. It should be mentioned that the choice of system utilized will be a function of economics and availability, in addition to environmental concerns.

Several comparative studies have been carried out on the amount of soil disturbance associated with cable, tractor and skidder harvesting operations in western North America (Wooldridge 1960, Dyrness 1965, 1967b, Ruth 1967, Megahan and Kidd 1972, Bockheim et al. 1975, Smith and Wass 1976). The general consensus of these studies was that cable systems caused less soil disturbance and compaction, due largely to a reduction in yarding disturbance and road construction, and that these systems could operate on much steeper slopes. The greater soil compaction associated with tractor or skidder operations generally occurs on skidtrails and temporary extraction roads, resulting in a concentration and channelization of runoff (Dyrness 1965, Campbell et al. 1973, Case and Donnelly 1979). In addition to creating erosional problems, severely compacted soil may be detrimental

to trees and seedlings (Hatchell et al. 1970, Moehring and Rawls 1970, Case and Donnelly 1979, Froehlich 1979, Raghavan et al. 1981, Wert and Thomas 1981).

There are obvious differences in both topography and wood volumes between western North America and New Brunswick which may limit the practicality, both from an economic and logistic standpoint, of some western cable logging systems in New Brunswick. Nevertheless, it was felt that a brief overview of these systems might provide some insight into "environmentally sound" alternatives which could, with modifications, be applied to the New Brunswick situation. The design and operation of skyline and high-lead cable systems has been described by Wooldridge (1960), Binkley (1965), Ruth (1967), Fredriksen (1970), Rothacher (1970b), Studier and Binkley (1974), Cottell et al. (1976), Sauder and Nagy (1977), B.J. Sauder (1978) and E.A. Sauder (1980, 1981, 1982). The skyline system is generally of a larger scale than the high-lead system and yarding distance (including lateral yarding distance) is therefore greater, as is the degree of lift applied to the yarded logs. As a result, the formation of many relatively evenly distributed, shallow skidtrails across the slope is characteristic of the skyline system while, in the high-lead system, disturbance is concentrated on a few narrow skidtrails which are generally deeper and are oriented in a downslope direction (Ruth 1967). These deep skidtrails may accelerate surface

erosion; however, it should be mentioned that skyline logging can also cause considerable soil disturbance on steep slopes if the logs are allowed to slide downslope while being dragged laterally (Ruth 1967). The main advantage of the skyline system in reducing soil disturbance (and subsequent erosional problems) is the minimization of road, trail and landing densities due to the greater yarding area covered on any one setup of equipment (Binkley 1965, Dyrness 1967b, Ruth 1967, Megahan and Kidd 1972). The reduction in the amount of required road may partially offset the greater yarding costs normally associated with the skyline system (Binkley 1965, Sauder and Nagy 1977). The maximum benefit of the skyline system, in terms of minimal soil disturbance, occurs when the logs are carried high enough to prevent or minimize contact with the ground (Binkley 1965, Dyrness 1967b, Ruth 1967, Megahan and Kidd 1972); it therefore works best on concave (Megahan and Kidd 1972) and uniform (Dyrness 1967b) slopes. Logs can even be transported across streams with a minimum of disturbance (Wooldridge 1960, Dunford 1962, Carson 1975).

The high-lead system is, however, the most common cable logging method in the coastal forests of British Columbia (Bockheim et al. 1975, Sauder and Nagy 1977), largely because it is more versatile (Sauder and Nagy 1977, B.J. Sauder 1978). The more complex skyline system can only be economically used on steep slope situations (B.J. Sauder

1978) and where log quality and size, and the volume of wood per unit area, is high (E.A. Sauder 1982). Frequent movement would also be expected to reduce the efficiency of the larger skyline system, although the utilization of a yarding crane as opposed to a fixed spar improves the mobility of this system (Sauder and Nagy 1977). The substitution of grapples for chokers in cable systems appears to be advantageous in some situations (E.A. Sauder 1980, 1981) but would probably be restricted to high volume logs.

It is obvious that the cable logging systems used in the western coastal forests would have to be reduced to a much smaller scale in order to operate economically in eastern North American forests due to the relative lack of steep terrain and reduced tree volumes, as well as the generally smaller land ownerships (Kochenderfer and Wendel 1980). Any cable system would have to be very mobile, particularly if it is being used to harvest within buffer zones. Several such types of cable logging systems, operating on a basically modified skyline technique, have been developed; due to the reduced scale of these systems the logs being winched often come into contact with the soil and top dragging is common. Hawkes (1979) provided a detailed description of many cable harvesting systems designed to handle small timber. Cottell et al. (1976) undertook an examination of six different cable systems for harvesting wood in interior British Columbia and Alberta and concluded that further

research and system refinement were necessary, although preliminary results were promising. Numerous tests (involving both clearcuts and partial cuts) of the URUS mobile yarder have been carried out in high value hardwood stands on slopes of the Appalachian Mountains in Virginia (Wendel et al. 1974, Gibson and Biller 1975, Kochenderfer and Wendel 1978, Patric and Gorman 1978, C. Smith 1978). In the case of partial cuts it was necessary to cut a 5 to 6 m wide corridor directly under the skyline (Wendel et al. 1974, Gibson and Biller 1975, Patric and Gorman 1978, C. Smith 1978). Lateral yarding distances for this URUS yarder were in the order of 23 to 30 m (Wendel et al. 1974, Kochenderfer and Wendel 1978, C. Smith 1978) while the length of area yarded ranged from 180 to 270 m (Wendel et al. 1974, Gibson and Biller 1975, Kochenderfer and Wendel 1978, C. Smith 1978). The general consensus was that, although the system required more advance planning than the conventional skidder operation and was more expensive for partially cut areas (Kochenderfer and Wendel 1978), it was relatively easy to set up (Wendel et al. 1974) and resulted in less soil disturbance through a reduction in the number of required roads and trails, particularly on steeper slopes. In partial cuts, damage to the residual stand was also reduced. A Smith Timbermaster cable logging operation with a lateral yarding distance of approximately 15 m has given satisfactory results in Newfoundland (Colbert 1980). In New Brunswick, a

RMS Ecologger cable system is currently being utilized by Fraser Inc. on steep terrain of the Restigouche River watershed (Corey 1981). The lateral yarding distance was reported to be from 15 to 25 m and, although more expensive, this system has been found to considerably reduce soil disturbance. A relatively high degree of mobility was achieved here as the Ecologger unit itself utilized previously constructed roads on ridgetops (which were used in a conventional harvesting operation) and a D-6 tractor served as a tailhold for the skyline cable (Corey 1981). Promising results have also been reported during testing of this Ecologger system in Newfoundland (Heidersdorf 1978). Yarding cranes, either truck-mounted (Kochenderfer and Wendel 1980) or self-moving units (Parker 1976), have also been used in harvesting operations in eastern North America.

The previously discussed cable logging systems have involved some degree of log lifting such that the butt end of the log, at least, did not normally drag on the ground while being winched to the landing. If harvesting is to take place within the buffer zone consideration should also be given to the ground-winching of logs out of the zone, particularly if some form of selection cutting is utilized. This was successfully done following selection cutting of a 10 to 20 m wide buffer zone in West Virginia (Aubertin and Patric 1974); no machinery was allowed in the zone and logs were winched to roads which were no closer

than 30 m to the stream. The idea of ground-winching with a cable is certainly not new (Garrison and Rummell 1951) and, although it undoubtedly creates more soil disturbance than the raised-log cable methods, it may be preferable to the formation of trails and roads within the buffer zone. Cable skidders, possibly equipped with extra cable, may have potential for this ground-winching of logs out of the buffer zone. However, C. Smith (1978) indicated that any additional pulling of the cable would result in additional costs and more work for the operator. Hoffman (1982) discussed the feasibility of using small, portable radio-controlled winches for woodlot harvesting in conjunction with self-releasing snatchblocks and cones to minimize pulling resistance and disturbance to the soil and residual vegetation. The ability of such small winches to economically handle large logs in a commercial operation requires further evaluation (D. Davies, pers. comm.). Nevertheless, the potential of utilizing a small winch to remove and prebunch logs from the riparian zone, instead of using a higher priced skidder fulltime, merits further consideration. Winches are currently available for mounting on farm tractors as well (Folkema 1977, Ryans 1980) but a small skidder with a winch would probably be more suitable for commercial harvesting within a riparian zone due to the unsuitability of farm tractors on rough terrain or on slopes greater than 10 to 15% (Folkema 1977, Falk 1979).

An alternative to winching logs out of the riparian zone utilizing only cables is to use equipment within the zone which will minimize soil disturbance. McMorland (1980) felt that the skidding of logs with small (63 to 78 horsepower) crawler tractors on slopes greater than 30% would be economical and would also normally reduce the amount of soil disturbance, as compared to larger tractors or wheeled skidders. On more level slopes such crawler tractors would not be as productive as skidders (McMorland 1980) due to their slower speed (Falk 1979, McMorland 1980) and higher maintenance costs (Falk 1979, Mellgren 1982) but still may cause less soil disturbance. Horse skidding is an alternative in areas where it is desirable to minimize soil disturbance; in addition horses are maneuverable on rough terrain and steep slopes (Falk 1979). Small forwarders such as the F4-Dion (Anonymous 1982b) or Krabat (Hermelin 1979) may also be useful but their limited capacity may make them impractical for large wood operations (D. Davies, pers. comm.). Froehlich (1978) reported minimal soil disturbance following logging in Oregon with a high-speed FMC skidder, which has a track-wheel torsion bar suspension system. However, the cost of this machine is relatively high (D. Black, pers. comm.). Mellgren (1982) found that a skidder equipped with high flotation tires caused less soil rutting and compaction, and was more productive, on boggy ground than a skidder equipped with ordinary tires. However, the

extra cost of these tires, associated modifications to the skidder and reduced machine maneuverability (Mellgren 1982) may reduce their desirability. Preliminary examination of well drained sites in New Brunswick that have been harvested with a Koehring Feller-Forwarder have indicated minimal mineral soil exposure (D. Black, pers. comm.). Thus, even large machinery may have application for harvesting within riparian zones, provided that ground bearing pressure is relatively low.

Proper planning of both the road and skidtrail network prior to harvesting with conventional equipment may, in many cases, be the most practical and satisfactory alternative in minimizing soil disturbance (Hornbeck 1968, Hornbeck et al. 1975, DePape and Collins 1977, Bradshaw 1979, Falk 1979, Froehlich et al. 1981). In Oregon, preplanning skidtrails in partial cut operations produced greater spacing (30 to 75 m apart) and a more uniform pattern of skidtrails which resulted in a reduction in overall soil compaction and disturbance (Bradshaw 1979, Froehlich et al. 1981). This wider spacing resulted in a greater amount of time spent winching (greater winching distance) than in an unplanned system, but in one study (Froehlich et al. 1981) this was offset by a reduction in the amount of time spent in positioning the skidder. Smith and Wass (1976) suggested that this planning would be most effective on moderate slopes (30 to 50%) while logging on steeper slopes

would likely necessitate a change in logging technique or equipment (e.g. smaller skidders). Directional felling of trees to facilitate winching to the skidtrails (Froehlich et al. 1981) and preset chokers (Bradshaw 1979) may improve productivity in this system. In at least some areas (Dickerson 1976, Froehlich 1979) the soil compaction produced from logging operations may be relatively long lasting; therefore, an attempt should be made to incorporate any existing skidtrails into the skidtrail network to minimize additional areas of soil compaction (Froehlich et al. 1981). The relationship of soil density with amount of vehicular traffic appears to vary with local conditions. Miles et al. (1981) found that considerable compaction of loam soil in California occurred on the first trip of a skidder or crawler tractor and suggested several recommendations to minimize overall compaction including the use of a low density skidtrail network, having skidtrails located on the driest soils, winching logs to the trail and using low ground pressure machinery. Froehlich (1978) found that the increase in soil density was largest during the first few trips of a FMC skidder and that sites with less litter cover and steeper slopes sustained more disturbance. However, Sidle and Drlica (1981) reported minimal soil compaction by the same machine type on low-use trails.

The type of cutting system can affect the amount of soil disturbance. Haupt (1960) found that group tree

selection caused less disturbance than single tree selection in Idaho due to the greater dispersion of skidtrails and roads required in the latter system. Full-tree skidding has been found to disturb more area and cause greater soil compaction than either tree-length or shortwood systems (Mace 1970, Zasada 1974). Recovery of soil compacted in the tree-length system may also be faster than in the full-tree system due to less initial compaction and the higher moisture content of the soil (slash reduces evaporation and retards advance growth in the cut area) (Mace 1971). However, results of soil compaction studies have been variable (Dickerson 1976) and Falk (1979) concluded that it was not certain whether full-tree or tree-length skidding caused more soil damage. Hermelin (1979) indicated that a shortwood forwarding system would require only a simple trail system and a low-density road network. Such a system is suited to cuts involving partial stand removal (e.g. thinnings) but, because it is relatively labor intensive, may be less feasible on large commercial operations (Hoffman 1982).

Burwell (1971) advocated uphill felling of trees, by attachment of a cable to each tree, to reduce the amount of debris in streams. However, such cable-assisted felling would only be economically feasible when large, valuable timber in steep terrain was involved (Dykstra and Froehlich 1976). In Oregon, Dykstra and Froehlich (1976) concluded that a buffer strip reservation (17 m), and associated loss

of the value of the timber on it, was in some cases more cost efficient than complete felling followed by cable yarding and cleaning of debris from the stream channel. The effect on other water quality parameters and the effect of debris removal were not considered.

Soil disturbance associated with the skidding of trees may be reduced in some cases if skidding only takes place under relatively dry soil conditions (Dyrness 1965, Zasada and Tappeiner 1969). Skidding may be restricted to ridgetops during periods of wet weather (Aubertin and Patric 1974). Soil compaction may be greater with uphill rather than downhill skidding (Sidle and Drlica 1981). Case and Donnelly (1979) suggested that skidder logging operations on poorly drained sites in Newfoundland should take place after freeze-up to avoid excessive soil compaction. The beneficial effect of winter logging operations in minimizing soil disturbance is well documented (Mace et al. 1971, Bockheim et al. 1975, Smith and Wass 1976, DePape and Collins 1977). Excessive amounts of snow may, however, impede mobility and result in high stumps (Smith and Wass 1976).

Sources of information regarding measures to prevent or reduce erosion problems in logging operations and road construction are numerous (Haupt 1959, Dunford 1962, Reinhart and Eschner 1962, Rice and Wallis 1962, Eschner and Larmoyeux 1963, Kidd 1963, Reinhart et al. 1963, Wallis 1963, Reinhart 1964, Haupt and Kidd 1965, Bethlahmy and Kidd

1966, Dyrness 1967a, 1970, Packer 1967a, 1967b, Kidd and Haupt 1968, Leaf 1970, Rothacher 1970b, Kimmins 1971, Larse 1971, Willington 1971, Megahan 1972, Rice et al. 1972, Swanston 1974, Zasada 1974, Wellburn 1975, Patric 1977, Rice 1977, Sidhu and Case 1977). More elaborate documentation and guidelines of "environmentally sound" timber harvesting and extraction processes are given by the United States Department of the Interior (1970), Rothwell (1971, 1978), L.W. Rowe (1974), Humar (1976), Hartung and Kress (1977), Megahan (1977), Case and Rowe (1978), L.W. Johnson (1979), Krag (1980), Yee and Roelofs (1980) and Talbot (1982), as well as the 1982 New Brunswick Forest Management Manual for Crown Lands. This is by no means an all-inclusive listing. In light of the amount of material available, it was deemed unnecessary to elaborate on this subject. A very brief summary of essentially "common sense" points taken from the above articles would include proper planning to minimize the road network and to ensure safe road and landing locations (from the standpoint of stream water quality), preventing heavy machinery from operating near streams, avoiding areas of high erosion risk, avoiding the excessive baring of mineral soil and properly constructing and maintaining roads.

This chapter has attempted to provide some insight into harvesting alternatives (both equipment and methods) in the event that harvesting within a buffer zone is permitted. Although selection cutting will normally require a greater

number of roads and trails for any given volume of wood removed than clearcutting (DePape and Collins 1977), the smaller degree of buffer zone alteration associated with selection cutting may make it more desirable. The majority of the equipment and methods that have been suggested are applicable to selection cutting and some may also be utilized in clearcutting.

Careful logging, along with proper location and minimization of roads and trails, appears to be very important in preventing increases in stream sediment levels. In most cases, any additional precautionary measures or use of specialized equipment or techniques need only apply to harvesting within the buffer zone itself. Once the wood is delivered to an area where logging activity is deemed not to have any influence on water quality more conventional methods can be utilized. Thus, any specialized equipment used for harvesting on steep slopes and/or buffer zones should be relatively mobile.

CHAPTER 7

SUMMARY

Buffer zones serve many purposes; they influence a number of water quality parameters, exhibit tremendous influence on stream ecology and fish populations, act as a deterrent to stream disturbance by logging equipment and can be valuable to wildlife. Additionally, they have aesthetic and recreational value (Streeby 1971, Miller et al. 1976, Nash 1977, King et al. 1979). The relative importance of each of these values will vary with the individual buffer zone. It has become clear during this literature review that the effect of logging disturbance within the buffer zone on any of these values is subject to a variety of interacting factors which in combination provide a degree of uniqueness to any particular watershed in terms of the importance of the buffer zone and the consequences of logging disturbance within it. This multitude of variables precludes quantitative identification of relatively easily obtained wood-related characteristics (e.g. basal area, volume per unit area, diameter limits) essential to the maintenance or enhancement of any fish or wildlife population. The combination of individual site characteristics such as plant species present, regeneration patterns, geology, soil type, drainage patterns, slope, aspect, climate and degree of disturbance to adjacent areas exerts a

tremendous influence on the outcome of any timber harvesting operation within the riparian zone. The tolerance levels and resiliency of fish and wildlife populations, being a function of such factors as stage of the life cycle, prior exposure, climate and degree of disturbance to adjacent watersheds and upslope areas, are also variable. Failure to uniformly quantify even these tolerance levels indicates that any identification of specific tree or stand parameters essential to fish and wildlife would be a gross over-simplification of a complex subject. Many adverse effects are rather short-lived and must be judged relative to their temporal occurrence in the life cycle of any particular organism. The indirect effects of disturbance are undoubtedly the most difficult to predict and are often the longest lasting.

In conclusion, harvesting of wood products within a buffer zone may be allowed if deemed environmentally safe; furthermore, no definitive buffer zone width should be established and rigidly enforced. A reasonable approach, as outlined in Chapter 3, would appear to be the establishment of "safe" limits such as those recommended by the New Brunswick Department of Natural Resources (1981), subject to alteration by the resource managers. This "alteration" would not always mean a reduction in the buffer zone width but could entail enlargement of the zone width in some cases to accommodate wildlife concerns (such as deer wintering

areas or extensively used transportation corridors), for aesthetic reasons or where the potential for stream ecosystem damage is deemed to be high. Careful consideration must be given to the major sources of stream water such as springs and small feeder streams. Logging operations must adhere to strict standards and, under no circumstances, should machinery be allowed to operate within 10 m of a stream except on a properly constructed road. If substantial water temperature increases are expected following tree removal, harvesting should be concentrated on the north side of east-west flowing streams.

It is felt that the feasibility and extent of timber harvesting within any buffer zone is a decision which must be made on a local level by a variety of resource personnel once all site characteristics have been evaluated and considered, including the condition of the buffer strip and its enhancement through management, as well the management possibilities for the local fish and wildlife populations. This report has attempted to present a review of possible factors associated with such a decision and the degree to which they will vary. Some of these factors have been given only periferal treatment in order to maintain a general overview while in other cases little information was available. It is hoped that this document will serve as an accessible source of information for any resource manager charged with the responsibility of riparian zone management.

CHAPTER 8

RESEARCH NEEDS

There is an urgent need to identify and manage buffer zones and to develop uniform decision-making processes within and among the various Forest Regions of the Province. It is proposed that numerous experimental timber harvesting operations involving a variety of site conditions be immediately established within recognized buffer zones in each of the administrative Forest Regions of New Brunswick. Although more detailed, long-range studies would provide better insight into specific parameters, they require time to establish a baseline of data from both the experimental and a control watershed prior to harvesting and the application of any results obtained has to be general in nature due to the variability reported to exist between watersheds.

Determination of the zones to be harvested, as well as the type and amount of harvest, will be based on the previously discussed considerations and will involve input from biologists and foresters. In an attempt to standardize the management of buffer zones a list of interim guidelines derived from the material presented in this report is presented in Appendix I. All management endeavors will be carefully evaluated from both an economic and environmental perspective. Based on the results of the experimental harvesting operations and management plans these guidelines

will be altered and refined. As no preliminary data will have been collected, analysis of the effect of any cutting operation on water quality and fish and wildlife populations will entail a substantial degree of subjectiveness. The amount of error encountered in this interpretation will be largely a function of the degree of cooperation between the resource personnel in each Forest Region. It is hoped that these experimental operations will provide the necessary experience to enable resource managers to make future rational decisions regarding harvesting in buffer zones based on regional differences in climate, desired or expected fish and wildlife population levels and the demand for wood. The coordination and dissemination of information and results between regions will take place from Fredericton.

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APPENDIX I

INTERIM GUIDELINES REGARDING THE MANAGEMENT OF BUFFER ZONES

1. No general prescription for buffer zone management can be made due to the tremendous variability in both the characteristics of these zones and the benefits provided by them. An individual decision regarding the management of each buffer zone is necessary and should be guided by the following recommendations, as well as the information contained in this report. The onus is on the local resource managers to integrate fisheries, wildlife and aesthetic values with the demand for wood fiber.

2. No evidence has surfaced to indicate that the present New Brunswick recommendation for stream (0.5 m or wider) protection by buffer zones should be altered. These are:

Slope	Minimum Zone Width
0-5%	15 m
5-10%	30 m
> 10%	60 m

These should be considered "safe" under most circumstances and should serve as a baseline from which the zone width may be reduced or expanded. The buffer zone should extend for the entire length of the stream.

3. It appears, from the standpoint of the fisheries resource, that the harvesting of timber within a buffer zone is feasible. A moderate reduction in canopy cover may not be unduly harmful to aquatic life and in some cases may increase stream productivity provided increases in water temperature, sedimentation and nutrient concentrations are minimized. In areas where the potential for stream ecosystem damage is deemed by the Regional Fisheries Biologist to be great, either due to natural conditions (e.g. steep slopes, highly erodible soils) or to inappropriate logging practices, cutting should be prevented. The following points should be considered.

(a) The amount of canopy removal necessary to produce excessive increases in water temperature will be variable depending on the individual stream (rate of flow, width, depth), topography (slope, aspect, valley depth) and additional water inputs, as well as the plant species present and their regeneration patterns. The amount of stream flow may be altered by cutting of the surrounding watershed. Small streams, with large surface areas relative to the volume of water and with low flows of water during the summer, are the most affected by riparian vegetation removal. The amount of merchantable vegetation which should be left along the stream will be a

function of the amount of unmerchantable vegetation (alders, shrubs, etc.) present.

(b) Prevention of increases in stream sedimentation should be a prime concern when timber is to be harvested from a buffer zone. Excessive amounts of stream sediment may have substantial adverse effects on fish populations. Measures designed to minimize sedimentation must be strictly adhered to. Soil disturbance and compaction must be minimized; this entails utilization of the proper equipment, as well as proper planning and care during the logging operation and extraction processes. Roads and landings should be conservative in number, well designed and located far enough away from streams to prevent stream sedimentation. Stream crossings by roads must be properly constructed and kept to a minimum; improper crossings may result in relatively long-lasting sediment sources. Logs shall not be skidded through streams, nor should equipment enter or cross streams. Machinery should not be allowed to operate within 10 m of any stream unless associated with a road. Adequate care should also be taken near small, intermittent streams or springs supplying substantial inputs of water to streams. Winter logging, logging after soil freeze-up or logging only under dry conditions should be given serious

consideration. On sites having high erosion potential (due to soil type, climate, slope, soil water content, etc.) harvesting within the buffer zone should be prohibited.

(c) Serious consideration should be given to the utilization of specialized equipment or techniques for the harvesting of wood within buffer zones. Winching of logs out of the zone, either on the ground or in the air, would eliminate the need for roads within the zone, as well as the compaction associated with machinery. Specialized equipment, as discussed in this report, may reduce the amount of site disturbance.

(d) Trees should not be felled in streams and logging debris should be prevented from entering streams. Only hand-clearing should be used in retrieving debris which does enter streams due to the potential for stream and stream bank disturbance by machinery. Burning of slash may result in additional stream sedimentation and should not normally take place.

4. The value of any particular buffer zone to wildlife, as well as the width and quality of zone required before benefits to wildlife are realized, will normally be harder to delineate than fisheries-related benefits. Nevertheless, the importance of these zones to wildlife

is very high and the wildlife biologist must have input in the formation of buffer zone management plans. It will be the responsibility of the Regional Wildlife Biologist to ensure a proper integration of disturbed and undisturbed buffer zones to maintain or enhance wildlife population levels in each local area. Zones known to receive high utilization either by species requiring undisturbed forest or as travel corridors should remain relatively intact, particularly if the surrounding watershed and neighboring buffer zones have been disturbed. Decisions regarding wildlife management in buffer zones will be tempered by regional variations in climate and importance placed on different wildlife species. For example, in areas having greater snow depths the utilization of buffer zones as deer wintering areas or travel corridors may be the prime consideration; additionally, any openings created within these zones may have a higher potential to impede travel due to the greater snow depths.

5. Selection cutting within the buffer zone will normally result in a less extreme modification of the zone than either partial cutting or clearcutting (in which removal of merchantable wood may be confined to patches or strips or may include the entire zone). Although the number of roads and skidtrails may be higher relative to

the amount of wood harvested, as compared to the clear-cut system, the amount of stream exposure may be less and the vegetative composition of the buffer zone will be relatively unchanged, resulting in little alteration of input of debris, invertebrates and other terrestrial matter into the stream ecosystem. The residual vegetation will retard sediment and nutrient increases in the stream water. Maintenance of the overall canopy cover in conjunction with increased browse production in small openings (e.g. group selection) may also be beneficial to wildlife species. Selection cutting may be advisable for aesthetic reasons as well.

6. The management of buffer zones to enhance and prolong the benefits afforded to fish and wildlife species, in conjunction with the harvesting of wood, should be the concern of and involve input from fisheries, wildlife and forest managers. A knowledge of the present importance of each particular zone and how an alteration will affect the residual vegetation (e.g. potential for blow-down) and the subsequent importance of the zone is required. Aesthetic values must also be considered. Although the discussion has dealt largely with streams, a similar approach to the management of buffer zones associated with rivers and lakes should also be taken. It is important to remember that the extra costs

associated with specialized equipment or techniques and the extra care and planning of the logging operation may be at least partially compensated for by the normally high value of wood found in buffer zones. Additionally, timber harvesting within buffer zones, necessitating a greater than normal trade-off with fisheries, wildlife and aesthetic values, will represent only a small portion of any annual harvest operation.

APPENDIX II

SUMMARY OF NASHWAAK EXPERIMENTAL WATERSHED PROJECT

The following is a brief summary of those research aspects of the Nashwaak Experimental Watershed Project which pertain in some way to the material discussed in this report. This information has been derived from the Annual Reports of the project and is based on the research of the following people: R.B.B. Dickison, D.A. Daugharty, Dr. H.H. Krause, H.B. King, Dr. H. Van Groenewoud, Dr. D.C. Eidt, J.H. Meating and J.D. Martin. To obtain more detailed information the Annual Reports or publications derived from specific avenues of research should be consulted.

This project involved a paired watershed approach with a calibration period for several parameters, followed by the clearcutting of one watershed. The Narrows Mountain Brook Watershed (391 ha) was, with the exception of a 60 m wide buffer strip left along each side of the mainstem of Narrows Mountain Brook, commercially clearcut between May 1978 and March 1979 using chainsaws and wheeled skidders in a tree-length operation. The Hayden Brook Watershed (660 ha) served as the undisturbed, control watershed.

An inherent danger with any project involving the experimental harvesting of a watershed is that additional care and precautionary measures may be taken beyond what would be realized in a normal harvesting operation.

Consequently, results may indicate a less than normal impact of the harvesting operation on any particular environmental aspect (e.g. sediment levels, stream invertebrate or fish populations). The Scientific Committee of the Nashwaak Experimental Watershed Project recognized this and an attempt was made to ensure that the experimental harvesting operation approximated normal harvesting operations as closely as possible (S. Hoyt, pers. comm.). Irregardless of this potential problem, such an experimental watershed project is additionally valuable in identifying the underlying mechanisms and problem areas, as well as illustrating which precautionary measures are environmentally and economically feasible.

A general lack of statistical testing was evident in the Annual Reports; this was undoubtedly influenced by the high degree of natural variability in the various measured parameters. Thus, any interpretation of results should be made in relation to the previously discussed material in an attempt to establish consistent trends.

The hydrological effects of this clearcutting operation appear to have followed a trend similar to that reported in the literature. There was an annual increase in the amount of discharge from Narrows Mountain Brook as well as an increase in summer peak flows, both of which appeared to decline by the second season following treatment. Snow

melt was earlier on the clearcut area resulting in a desynchronization of runoff from the two watersheds.

Suspended sediment and nutrient concentrations in Narrows Mountain Brook also followed expected patterns. Increases in stream sediment concentrations were usually associated with specific sources such as roads, stream crossings, culverts, etc. and occurred following storm events and spring freshets. By the second year following the cutting operation there was a reduced occurrence of suspended sediment increases; however extreme storm events still produced relatively high sediment levels. Increases in the nutrient content of Narrows Mountain Brook were small and generally insignificant for sodium, potassium, calcium and magnesium, although the net loss of potassium from the watershed was found to be relatively great. However, a substantial increase was seen in the stream water nitrate concentrations and losses of nitrogen from the clearcut watershed continued to increase two years after the treatment. Increased nitrification was noted in soils which previously supported hardwood species while no change was seen in the relatively poorly drained soils previously associated with softwood species. The increased concentration of phosphorus in the steam water was considered to be a result of increased stream sedimentation.

High variability in water temperature trends and levels was reported for the Narrows Mountain Brook. As

expected the higher water temperature readings (up to 21°C) were associated with riparian vegetation removal.

No detectable change was noted in the decomposition rate of allochthonous detritus in the stream before, during, or after the clearcutting operation.

An increase in the proportion of fine sand particles was first reported on an upper stretch of the mainstem of Narrows Mountain Brook, downstream from the point where the riparian strip began. By the fall of 1979 larger amounts of fine sand were also noted in the lower portion of the mainstem. Although analysis of the stream invertebrate population was incomplete it was suggested that the filling of interstices in gravel and rubble areas by this fine sand could produce an invertebrate species alteration. In addition, the amount of mud on the bottom of both branches of Narrows Mountain Brook increased substantially following the cutting operation. It seems likely that this could also have a pronounced effect on the invertebrate community.

Electrofishing data from Hayden Brook and the buffer strip and clearcut sites of Narrows Mountain Brook revealed no marked changes in fish populations attributable to the cutting operation or removal of riparian vegetation. The fish populations were found to exhibit considerable natural fluctuation.